Integrated Disposal Facility Risk Assessment

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EXECUTIVE SUMMARY

A. SUMMARY

Various operations at the U.S. Department of Energy's (DOE) Hanford Site in south central Washington State have produced low-level radioactive waste (some of which are mixed with hazardous chemicals). The two DOE Field Offices at the Hanford Site are evaluating options for disposing these wastes. One major alternative being considered is to dispose of all low-level waste other than that generated during environmental remediation actions in an Integrated Disposal Facility (IDF) near the PUREX plant in Hanford's 200 East Area starting in fiscal year 2006.

According to Hanford's *Integrated Mission Acceleration Plan*[1], a performance risk assessment for the IDF is to be performed by June 12, 2003. This performance risk assessment uses the data, methods, and knowledge from earlier performance assessments for the disposal of low-level wastes at various locations. If DOE selects the alternative analyzed in this document, the 2001 Immobilized Low-Activity Waste (ILAW) Performance Assessment[2] will be updated as required by the DOE Order on Radioactive Waste Management (DOE O 435.1)[3].

This assessment shows that the performance objectives defined in this document (which are based on the appropriate and relevant regulations) should be met with a reasonable expectation with the disposal of waste planned for the IDF.

B. SOURCES OF WASTE

The candidate low-level waste that may be disposed of at the IDF can be classified into four (4) categories:

- **Low-level waste (LLW) -** waste that contains man-made radionuclides but which is not classified as high-level waste or transuranic waste. This waste could have been generated on the Hanford Site or could have been imported from offsite. Category 1 (unstabilized) waste has the lowest level of radionuclides. Category 3 (stabilized) waste has higher concentrations and/or amounts and is grouted before disposal.
- **Mixed low-level waste (MLLW)** - waste that contains man-made radionuclides but which is not classified as high-level waste or transuranic waste and which contains materials that are regulated under RCRA or the

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[1] RPP-13678, *Integrated Mission Acceleration Plan*, RPP-13678, Revision 0, CH2M Hill Hanford Group, Inc., Richland, Washington, March 2003.

Deliverable 3 of MAAP 2.5 (Integrate LAW/LLW/MLLW Disposal Options) is "Prepare a Performance Risk Assessment of the integrated disposal facility" and is due June 12, 2003.

[2] F.M. Mann, K.C. Burgard, W.R. Root, R.J. Puigh, S.H. Finfrock, R. Khaleel, D.H. Bacon, E.J. Freeman, B.P. McGrail, S.K. Wurstner, and P.E. LaMont, 2001, *Hanford Immobilized Low-Activity Waste Performance Assessment: 2001 Version*, DOE/ORP-2000-24, Department of Energy Office of River Protection, Richland, Washington.

[3] "Radioactive Waste Management", DOE O 435.1, U.S. Department of Energy, Washington, D.C., July 9, 1999.

corresponding dangerous waste management laws of the State of Washington.

- **Immobilized Low-Activity Waste (ILAW)** - Hanford tank waste that has undergone separations treatment to remove the bulk of the radionuclides and then solidified at the Hanford Waste Treatment and Immobilization Plant (WTP). Presently, the only DOE-approved solidification process is WTP vitrification.
- **Failed or Decommissioned Melters** - High-level and low-activity waste melters used to treat tank waste in the WTP.

C. PERFORMANCE OBJECTIVES

The performance objectives for this risk assessment are those proposed for the 2005 Immobilized Low-Activity Waste Performance Assessment as documented in *Performance Objectives for the Hanford Immobilized Low-Activity Waste (ILAW) Performance Assessment*[4]. They are based on evaluating all federal and State of Washington relevant and appropriate laws and regulations.

The most significant performance objectives are

- The all-pathways dose objectives of 25 mrem effective dose equivalent (EDE) in a year
- The drinking water dose objectives for beta and gamma emitters of 4 mrem EDE in a year
- The incremental lifetime cancer risk due to chemicals
- The inadvertent intruder all-pathways chronic dose objectives for a post-driller resident of 100 mrem EDE in a year.

The first three objectives are evaluated at a point 100 meters down gradient from the disposal trench and for times of 1,000 and 10,000 years after closure. The last objective is evaluated at the disposal facility at 500 years (consistent with earlier Performance Assessments[2,5,6]).

D. APPROACH AND MAJOR DATA SOURCES

This risk assessment uses the data, methods, and knowledge of previous performance assessments that have analyzed the disposal (actual or planned) of the wastes in disposal configurations that differ from the integrated disposal facility concept. There have been two major efforts:

- **Solid Waste Burial Grounds** - In the mid-nineties, the *Performance Assessment for the Disposal of Low-Level Waste in the 200 West Area Burial Ground*[5] and the *Performance Assessment for the Disposal of*

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[4] F. M. Mann, 2002, *Performance Objectives for the Hanford Immobilized Low-Activity Waste (ILAW) Performance Assessment*, RPP-13263, CH2M Hill Hanford Group, Inc., Richland, Washington.

[5] M. I. Wood, R. Khaleel, P. D. Rittmann, A. H. Lu, S. H. Finfrock, R. J. Serne, and K. J. Cantrell, 1995, *Performance Assessment for the Disposal of Low-Level Waste in the 200 West Area Burial Grounds*, WHC-EP-0645, Westinghouse Hanford Company, Richland, Washington.

Low-Level Waste in the 200 East Area Burial Ground[6] were completed and approved by DOE. These performance assessments have been maintained with the most recent annual summary submitted in September 2002[7].

- **ILAW Disposal Facility (Project W-520)** - The first performance assessment was prepared and approved in 1998. The current performance assessment (*Hanford Immobilized Low-Activity Waste Performance Assessment: 2001 Version*[2] was also approved. This performance assessment is also being maintained with the most recent annual summary issued in August 2002[8].

Information on the IDF configuration was generated based on the on-going detailed design process. Inventory values for radionuclides and chemicals for various waste types as well as waste form release data and methods are based on the prior performance assessment efforts and their related activities. Geologic, hydrologic, and geochemistry data as well as the methods for flow and transport simulation are also based on prior ILAW performance assessment activities.

E. SUMMARY OF RESULTS

All performance objectives associated with release and migrations of radionuclides through the groundwater pathway to the point of compliance are met with a wide margin (ratio of performance objective to estimated impact [factor of ~6]). The performance goals associated with release and migration of hazardous chemicals to the point of compliance are met with an even wider margin (factor of ~8) than met by radionuclides. The intruder dose performance objective is met with a smaller margin (factor of ~4).

1. Groundwater Impacts

The contaminant breakthrough curves for groundwater impacts of the three main categories of waste (Category 1 solid waste, Category 3 solid waste, and ILAW glass) have different temporal distributions, as seen from Figure ES-1. The impacts from Category 1 wastes, which have quick releases, peak early (at ~2,400 years after facility closure for contaminants with K_d = 0 mL/g) and are insignificant after a few more thousand years. The impacts from Category 3, which are encased in grout, peak a bit later but in the same general time frame as Category 1 wastes. However, because of the continued release from Category 3 wastes, impacts are still significant at the longest times calculated (20,000 years after facility closure). The impacts from glass are

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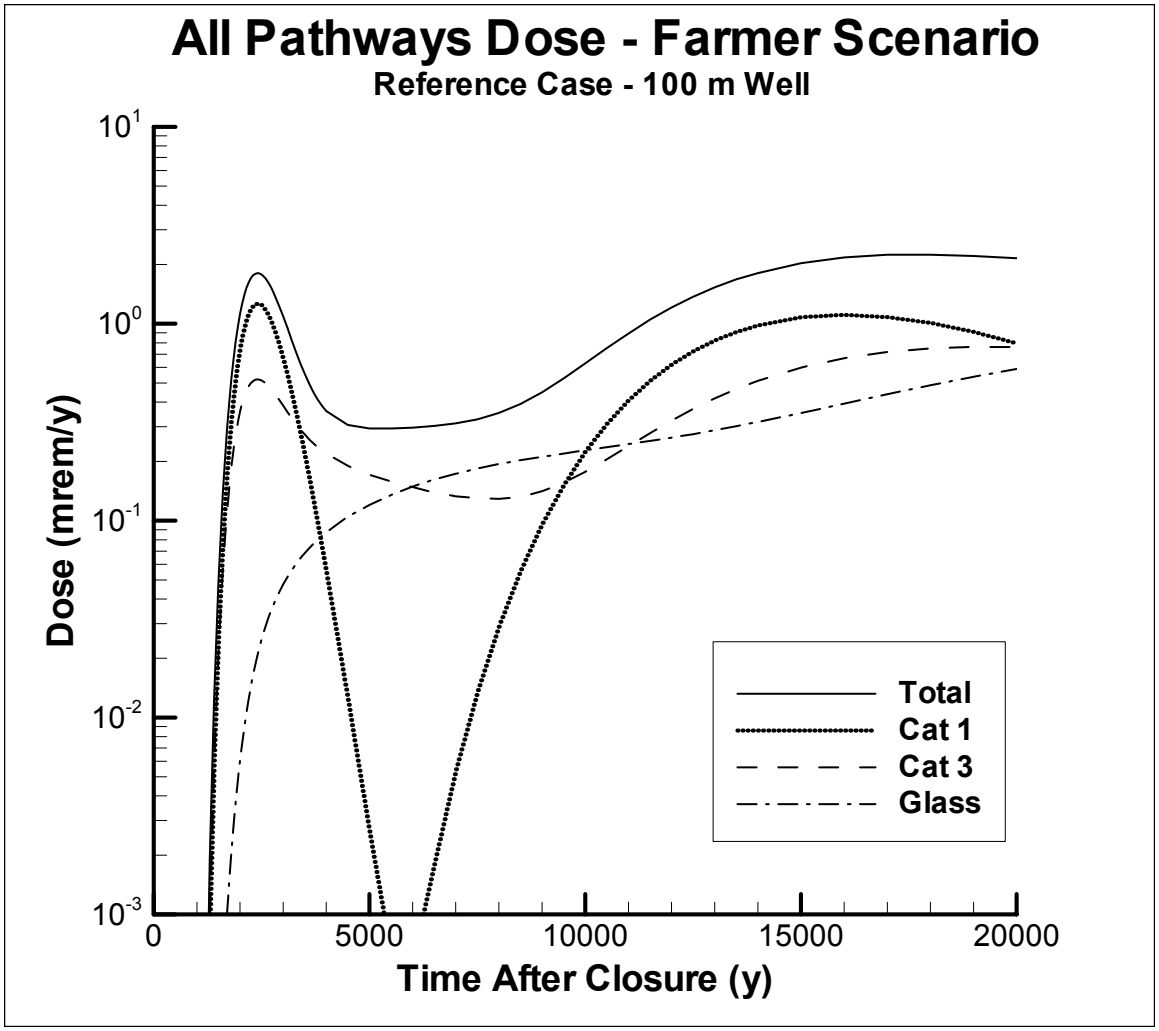
[6] M. I. Wood, R. Khaleel, P.D. Rittmann, S.H. Finfrock, T.H. DeLorenzo, and D.Y. Garbrick, 1996, *Performance Assessment for the Disposal of Low-Level Waste in the 200 East Area Burial Grounds*, WHC-EP-0875, Westinghouse Hanford Company, Richland, Washington.

[7] M. I. Wood, 2002, *Performance Assessment Review Report, 2001-2002, Annual Review of the 200 West and 200 East Area Performance Assessments*, (letter to Mr. Michael H. Schlender , Acting Manager, Richland Operations Office, U.S. Department of Energy, letter #FH-0204558, dated September 30), Fluor Hanford, Richland, Washington.

[8] F.M. Mann, 2002, *Annual Summary of ILAW Performance Assessment for 2002*, DOE/ORP-2000-19, Revision 2, Office of River Protection, U.S. Department of Energy, Richland, Washington.

insignificant at the times when the mobile contaminant impacts from Category 1 or 3 wastes peak, but the impacts plateau for longer times (greater than 4,000 years after facility closure).

Figure ES-1. Time Dependence of the Estimated Farmer Scenario All-Pathways Dose at a Well 100 m Downgradient from the Disposal Facility.



The peak groundwater impacts are due to Category 1 waste. The impacts from Category 1, Category 3 and glass wastes are comparable at 10,000 years. Because only a relatively few Category 1 packages are expected to drive the results (i.e., those packages with high technetium/iodine content), the amount of Category 1 waste accepted is manageable (e.g., these wastes can be disposed as Category 3 waste, if necessary). Impacts from melter disposal are not significant relative to impacts from other wastes.

The contaminant breakthrough impacts for the groundwater for ILAW-glass disposal are about five times higher than those presented in the base analysis case of the *Hanford Immobilized Low-Activity Waste Performance Assessment: 2001 Version*[2], but still far below performance objectives. The key drivers are increased Tc-99 inventory

due to the removal of the technetium separations process from WTP[9], decreased groundwater dilution due to the placement of the disposal trenches further towards the southern end of the disposal site, and the decrease in contaminant release due to the size of the containers. Additional analyses and assumptions that could reduce the estimated impacts (such as the estimated impacts from a two-dimensional modeling of the near-field (compared to one-dimensional modeling) and better waste form performance) have not been included in this analysis.

The contaminant breakthrough impacts for the groundwater pathway for solid waste disposal are similar to those presented in the latest annual summary[7]. A straightforward comparison with the burial ground analysis is not plausible because several key assumptions affecting estimated impacts are different, leading to both increases and decreases in these estimates. For example, hydrogeologic properties of the unconfined aquifer at this site versus the 200 West Area site create a larger dilution effect and lower the estimated impacts. However, in both cases, performance objectives are easily satisfied.

2. Protection of General Public

The estimated all-pathways doses are significantly lower than the performance objectives during the first 10,000 years. At the DOE time of compliance (1,000 years) the estimated impact is insignificant.

Table ES-1. Comparison of Estimated Impacts with Performance Objectives for Protecting the General Public. The DOE time of compliance is 1,000 years.

Performance Measure	Performance Objective	Estimated Peak Impact During First 1,000 years [(a)]	Estimated Peak Impact During First 10,000 years [(b)]
All-pathways [mrem in a year]	25.0		
Farmer Scenario		1.2×10^{-10}	1.8
Residential Scenario		0.73×10^{-10}	1.1
Industrial Scenario		0.22×10^{-10}	0.32
Incremental Lifetime Cancer Risk (Chemicals)*	10^{-5}	7.9×10^{-17}	5.6×10^{-7}
Hazard Index (Chemicals)*	1.0	1.8×10^{-11}	0.12

*Based on chromium, nitrate, and uranium inventory
[(a)] Peak impacts occur at the end of the 1,000 year period
[(b)] Peak impacts occur at about 2,400 years after closure

The greatest contributors to the peak all-pathways dose are mobile contaminants from the Category 1 wastes, which peak in the few thousand-year time frame (see Figure ES-1). Category 3 wastes show a peak at about the same time. For times exceeding 10,000 years, the contributions from the mobile contaminants from glass, contaminants

[9] R. J. Schepens, 2003, Contract No. DE-AC27-99RL 140 47 - National Environmental Policy Act (NEPA) Support Referencing Approval of Bechtel National Inc. (BNI) Trend TN-24590-02-00666, Removal of Technetium Ion Exchange System, (letter to Mr. E.S. Aromi, Manager, CH2M Hill Hanford Group, Inc., letter CTS No. 0300480, dated February 19), U. S. Department of Energy - Office of River Protection, Richland, Washington.

from Category 3 wastes, and the slightly retarded contaminants from Category 1 wastes are comparable.

Up to about 5,000 years, the major contributors to the farmer scenario all-pathways estimated dose are I-129 (~90%) and Tc-99 (~10%). At 10,000 years, Np-237 contributes 44% of the all-pathways dose, Tc-99 contributes 35%, I-129 contributes 17%, and other radionuclides contribute 4%.

3. Protection of the inadvertent intruder

The inadvertent intruder impacts are displayed in Table ES-2. The time of compliance for protecting the inadvertent intruder starts at 500 years after closure. The acute exposure performance objective is met by a factor of ~500. The maximum acute exposure dose is based on all the exhumed waste being ILAW. ^{126}Sn is the most important radionuclide. The continuous exposure performance objective is met by a factor of approximately four. The maximum homesteader dose is based on all the exhumed waste being LLW/MLLW. ^{241}Pu, ^{243}Am, and ^{239}Pu are the major contributors.

The estimated impacts for the inadvertent intruder can be mitigated through operational controls based on projected container inventories. Such operational controls will be better defined as the project matures.

Table ES-2. Comparison of Estimated Impacts with Performance Objectives for Protecting the Inadvertent Intruder. The time of compliance starts at 500 years.

Performance Measure	Performance Objective	Estimated Impact at 500 years
Acute exposure [mrem]	500.0	1.06
Continuous exposure [mrem in a year]	100.0	26.8

4. Protection of groundwater resources

Table ES-3 compares the estimated impacts to the performance objectives for protecting the groundwater resources. At the DOE time of compliance (1,000 years) and the point of compliance (at a well 100 m downgradient of the disposal facility), the groundwater impacts are not significant. For the first 10,000 years the estimated impacts are approximately a factor of six less than the performance objectives for beta-photon emitters and a factor of 150 less than the performance objectives for the alpha-emitting radionuclides for the reference case. The concentration of radium is insignificant.

The most important isotopes are the same as those for the all-pathways scenario.

Table ES-3. Comparison of Estimated Impacts with Performance Objectives for Protecting Groundwater. The DOE time of compliance is 1,000 years.

Performance Measure	Performance Objective	Estimated Impact at 1,000 years [(a)]	Estimated Peak Impact for the first 10,000 years [(b)]
βγ Emitters [mrem/year]	4.0	4.7×10^{-11}	0.70
Alpha-emitters [pCi/L]	15.0 [(c)]		
All radionuclides		0 [(d)]	0.19
Non-uranium radionuclides		0 [(d)]	0.10
Ra [pCi/L]	5.0	0.0 [(d)]	0.0 [(d)]

[(a)] Peak impacts occur at the end of the 1,000-year period.

[(b)] Peak impacts occur at about 2,400 years after closure

[(c)] The performance objective excludes uranium contribution to the concentration

[(d)] The estimated impact at 1,000 years after facility closure was less tan 1×10^{-20} pCi/L

5. Protection of Air Resources

Table ES-4 compares the estimated impacts to the performance objectives for protecting air resources. The DOE time of compliance is 1,000 years and the point of compliance is just above the disposal facility. The estimated impacts are lower than the performance objectives and are based on extremely conservative assumptions.

Table ES-4. Comparison of Estimated Impacts with Performance Objectives for Protecting Air Resources. The DOE time of compliance is 1,000 years. The point of compliance is just above the disposal facility.

Performance Measure	Performance Objective	Estimated Impact at 1,000 years
Radon [pCi m^{-2} $second^{-1}$]	20.0	2.7
Other radionuclides (^{3}H and ^{14}C) [mrem in a y]	10.0	0.44

F. PERFORMANCE SENSITIVITY TO KEY PARAMETER UNCERTAINTIES

The key uncertainties of this analysis are as follows:

- Uncertainties in inventory
- Uncertainties in release rates from Category 3 and ILAW
- Uncertainties in retardation for slightly retarded contaminants from Category 1 waste
- Uncertainties in recharge
- Uncertainties in groundwater flow.

The greatest groundwater pathway impacts are from Category 1 and Category 3 solid waste disposal. The inventory for these wastes is quite uncertain since they depend on future decisions. In particular, the amount of offsite waste to be disposed at Hanford as a result of the Solid Waste EIS Record of Decision is uncertain. Better estimates of inventory values for WTP secondary waste streams are expected as the WTP contractor finishes design and as operations begin.

For long time periods (i.e., over 5,000 years), the impacts are sensitive to the release rates from Category 3 wastes and from ILAW. The release rate from Category 3 waste was estimated based on a representative diffusion coefficient. The use of an effective diffusion model to represent the release rate of contaminants from grouted LLW and MLLW needs to be investigated further. Work is continuing on ILAW release rates and as shown by the latest ILAW annual summary[8], the ILAW release rates used here are conservative.

Interestingly, the slightly retarded contaminants from Category 1 waste have similar estimated impacts when compared to the mobile contaminants from Category 3 wastes and ILAW. The retardation factor for the slightly retarded contaminants is based on the lowest values thought to be likely in the Hanford environment. More realistic values for retardation would lower the estimated impacts.

Although not explicitly modeled in this document, the 2001 ILAW PA[2] showed the strong dependence of estimated impacts on the rate at which moisture infiltrates the ground surface and subsequently enters the disposal facility (i.e., the recharge rate). Again, conservative values were used in this analysis. Better estimates should lower estimated impacts.

G. SUMMARY

All of the estimated impacts easily meet the performance objectives. The estimated all-pathways dose, beta-photon drinking water dose, and concentration of alpha-emitting radionuclides in groundwater for the reference case are more than a factor of six (6) lower than the corresponding performance objective during the first 10,000 years after facility closure (2046). This margin increases by many orders of magnitude for the time of compliance of 1,000 years, as the travel time through the vadose zone is longer than 1,000 years. These estimates are based on conservative assumptions and hence should provide reasonable expectation that human health and the environment will be protected.

The most significant change from the 2001 ILAW performance assessment[2] is the inclusion of solid waste. Although the total inventories in this analysis are not significantly higher than analyzed in the 2001 ILAW performance assessment, the release rates of the solid waste are very much higher. Such higher release rates result in higher impacts than shown in the 2001 ILAW performance assessment. However, such impacts are consistent with impacts estimated in the most recent annual summary of the solid waste performance assessment.[7]

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LIST OF ACRONYMS

AEA	Atomic Energy Act
BBI	Best Basis Inventory
BNFL	British Nuclear Fuels, Limited
BNI	Bechtel National, Inc.
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFEST	coupled fluid, energy, and solute transport (code)
CFR	code of Federal regulations
CR	Columbia River
DOE	U. S. Department of Energy
Ecology	Washington State Department of Ecology
EDE	effective dose equivalent
EIS	environmental impact statement
EPA	U. S. Environmental Protection Agency
ERDF	Environmental Remediation Disposal Facility
HDPE	high-density polyethylene
HFSUWG	Hanford Future Sites Uses Working Group
HI	hazard index
HLW	high-level waste
HSRAM	Hanford Site risk assessment methodology
IDF	Integrated Disposal Facility
IHLW	immobilized high-level waste
ILAW	immobilized low-activity waste
ILCR	incremental lifetime cancer risk
IRIS	Integrated Risk Information System
K_d	distribution coefficient
LAW	low-activity waste
LAWABP1	sample name of glass used in analysis
LCRS	leachate collection and recovery system
LDR	land disposal restriction
LLW	low-level waste
MLLW	mixed low-level waste
NA	not available
NC	not calculated
NEPA	National Environmental Policy Act
ORP	Office of River Protection
PA	performance assessment

PEIS	preliminary environmental impact statement
PNNL	Pacific Northwest National Laboratory
PUREX	Plutonium-Uranium Extraction (facility)
RCRA	Resource Conservation and Recovery Act
rem	roentgen equivalent man (unit used for measuring effective dose of radiation
RL	Richland Operations Office
RPP	River Protection Project
STORM	subsurface transport over reactive multiphases
SWBG	Solid Waste Burial Ground
SWIFT	Solid Waste Integrated Forecast Technical
SWITS	Solid Waste Information Tracking System
SWPEIS	Solid (Radioactive and Hazardous) Waste Program Environmental Impact Statement
TWRS	Tank Waste Remediation System
TWRSO&UP	Tank Waste Remediation System Operation and Utilization Plan
USC	underground soil contact
VAM3DF	variably saturated analysis model code
VSL	Vitreous State Laboratory
WHC	Westinghouse Hanford Company
WIF	well intercept factor
WTP	Waste Treatment and Immobilization Plant

1.0 INTRODUCTION

1.1 PURPOSE

Various operations at the U.S. Department of Energy's (DOE) Hanford Site in south central Washington State have produced low-level radioactive waste (some of which are mixed with hazardous chemicals). The two DOE Field Offices at the Hanford Site are evaluating alternatives for weighing the various options in disposing these wastes. One major alternative being considered is to dispose of all low-level waste other than that generated during environmental remediation actions in one integrated facility in Hanford's 200 East Area starting in fiscal year 2006.

According to Hanford's *Integrated Mission Acceleration Plan (RPP 2003)*, a performance risk assessment for the Integrated Disposal Facility (IDF) is to be performed by June 12, 2003. This performance risk assessment uses the data, methods, and knowledge from earlier performance assessments for the disposal of low-level wastes at various locations including IDF. If DOE selects the alternative analyzed in this document, the 2001 Immobilized Low-Activity Waste (ILAW) Performance Assessment (PA) (Mann et al. 2001) will be updated as required by the DOE Order on Radioactive Waste Management (DOE O 435.1 [DOE 1999b]).

1.2 BACKGROUND

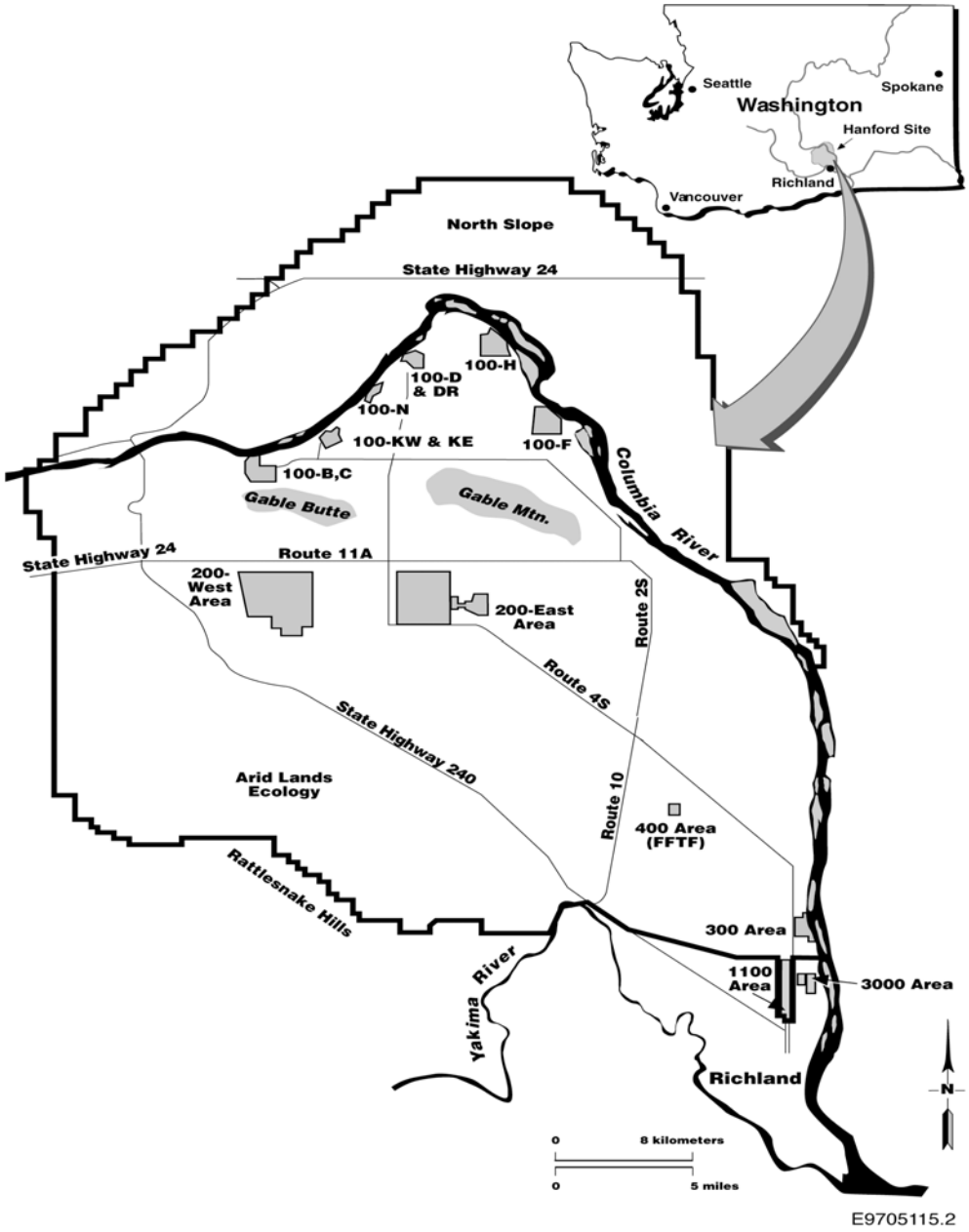
1.2.1 Hanford Site Background

The Hanford Site, in south-central Washington State (Figure 1-1), has been used extensively for producing defense materials by DOE and its predecessors, the U.S. Atomic Energy Commission and the U.S. Energy Research and Development Administration. Starting in the 1940's, Hanford Site operations were dedicated primarily to producing nuclear weapons materials. In the 1960's, operations were expanded to producing electricity from a dual-purpose reactor, conducting diverse research projects, and managing waste. In the late 1980's, the Site's original mission ended. This mission left a large inventory of radioactive and mixed waste (~55 million gallons) stored in 149 single- and 28 double-shell underground tanks in the Hanford Site 200 Areas. In addition, in operating the Hanford Site, large amounts of low-level radioactive waste have been generated. Finally, the Hanford Site is one of two DOE sites selected as disposal sites for low-level wastes generated elsewhere in the DOE complex (65 FR 10061)

Today, the Site's missions are environmental restoration, energy-related research, and technology development. As part of its environmental restoration mission, DOE is proceeding with plans to permanently dispose of the waste stored onsite. These plans are based on Revision 6 of the *Hanford Federal Facility Agreement and Consent Order* (Tri-Party Agreement) (Ecology et al. 1998-1) as well as the records of decisions arising from two current National Environmental Policy Act actions (Hanford Site Solid (Radioactive and Hazardous) Waste Program Environmental Impact Statement (SWPEIS) [DOE/EIS-0286D 2003] and Environmental Impact Statement for Retrieval, Treatment,

and Disposal of Tank Waste and Closure of Single-Shell Tanks at the Hanford Site [68 FR 1052]).

Figure 1-1. The Hanford Site and its Location in Washington State.



1.2.2 Regulatory Structure

The wastes at the Hanford Site are governed by three basic laws:

- The Atomic Energy Act (AEA)(AEA 1954), which covers radioactive wastes,

- The Comprehensive Environmental Recovery and Liability Act (CERCLA), which covers the wastes created from the environmental remediation of a facility, and
- The Resource, Conservation, and Recovery Act (RCRA), which covers the waste that contain hazardous chemical wastes (whether in treatment, storage, or disposal facilities). The State of Washington has been given authority over these mixed wastes (known in the State of Washington as dangerous wastes) by the U.S. Environmental Protection Agency.

A waste disposal action can be governed by one or more of these laws. Under the option being considered here, all wastes that are regulated under CERCLA would be disposed of at the Environmental Remediation Disposal Facility, which is located just east of Hanford's 200 West Area and all other waste will be disposed of at the IDF, which is located in south central part of Hanford's 200 East Area. The waste to be disposed of at the IDF can be grouped into 4 categories:

- **Immobilized Low-Activity Waste (ILAW)** - Hanford tank waste that has undergone separations treatment to remove the bulk of the radionuclides and then solidified at the Hanford Waste Treatment and Immobilization Plant (WTP). Presently, the only DOE-approved WTP solidification process is vitrification.
- **Failed or decommissioned Melters** - melters, possibly containing vitrified waste that is no longer needed by WTP.
- **Low-level waste (LLW) -** waste that contains man-made radionuclides but which is not classified as high-level waste or transuranic waste. This waste could have been generated on the Hanford Site or could be imported from offsite.
- **Mixed low-level waste (MLLW)** - LLW waste that also contains regulated materials under RCRA or the corresponding dangerous waste management laws of the State of Washington. Both ILAW and Failed or decommissioned Melters are considered MLLW. However, they are treated separately in this analysis, because of their extremely different waste forms.

1.3 PERFORMANCE OBJECTIVES

The performance objectives for this risk assessment are those proposed for the 2005 Immobilized Low-Activity Waste Performance Assessment as documented in *Performance Objectives for the Hanford Immobilized Low-Activity Waste (ILAW) Performance Assessment* (Mann 2002a). These are very similar to those used in the 2001 ILAW performance assessment (Mann 1999a) and in the solid waste burial grounds performance assessments (Wood et al. 1995a and Wood et al. 1996). The radiological objectives are displayed in Table 1-1. Since it is expected that radiological impacts will dominate, chemical objectives are abbreviated to cover only nitrate, chromate and the chemical aspects of uranium.

Table 1-1. Performance Objectives from the ILAW PA Used in this Risk Assessment.

Protection of General Public and Workers [a, b]	
All-pathways dose from only this facility	25 mrem in a year [c,d]
All-pathways dose including other Hanford Site sources	100 mrem in a year [e,d]
Chemical Carcinogens (Incremental Lifetime Cancer Risk)	10^{-5} [c,f]
Non cancer-cause chemicals (hazard index)	1 [c,f]
Protection of an Inadvertent Intruder [a, g, h]	
Acute exposure	500 mrem
Continuous exposure	100 mrem in a year
Protection of Groundwater Resources [a,b,e,i]	
Alpha emitters	
^{226}Ra plus ^{228}Ra	5 pCi/P
All others (excluding uranium)	15 pCi/P
Beta and photon emitters	4 mrem in a year
Nitrate	10. mg/liter
Cr^{6+}	0.1 mg/liter
Uranium	0.030 mg/liter
Protection of Surface Water Resources [a, b, j]	
Alpha emitters	
^{226}Ra plus ^{228}Ra	0.3 pCi/P [i]
All others (excluding uranium)	15 pCi/P [i]
Beta and photon emitters	1 mrem in a year [k]
Protection of Air Resource [b, h, l]	
Radon (flux through surface)	20 pCi m^{-2} s^{-1}
All other radionuclides	10 mrem in a year

[a] All doses are calculated as effective dose equivalents; all concentrations are in water taken from a well. Values given are in addition to any existing amounts or background.
[b] Evaluated for 1,000 years, but calculated to the time of peak or 10,000 years, whichever is longer.
[c] Evaluated at the point of maximal exposure, but no closer than 100 meters (328 feet) from the disposal facility. Also calculated 1 kilometer from the facility and just before groundwater enters the Columbia River.
[d] Main driver is DOE Orders on *Radioactive Waste Management* (DOE 1999b).
[e] Evaluated at the edge of 200 Area Core Zone (assumed to be 1 kilometer from the disposal facility).
[f] Main driver is Washington State Model Toxics Control Act.
[g] Evaluated for 500 years, but calculated from 100 to 1,000 years.
[h] Evaluated at the disposal facility.
[i] Main driver is National Primary Drinking Water Regulations.
[j] Evaluated at the Columbia River, no mixing with the river is assumed.
[k] Main driver is Washington State Surface Water Standards.
[l] Main driver is National Emission Standards for Hazardous Air Pollutants.

1.4 APPROACH AND MAJOR DATA SOURCES

This risk assessment uses the data, methods, and knowledge of the performance assessments that have analyzed the disposal (actual or planned) for the wastes to be disposed in the integrated disposal facility. There have been two major efforts:

- Solid Waste Burial Grounds. In the mid-nineties, the *Performance Assessment for the Disposal of Low-Level Waste in the 200 West Area Burial Grounds* (Wood et al. 1995a) and the *Performance Assessment for the Disposal of Low-Level Waste in the 200 East Area Burial Ground* (Wood et al. 1996) were created and approved by DOE. These performance assessments have been maintained with the last annual summary being submitted in September 2002 (Wood 2002).
- ILAW. The first performance assessment was created and approved in 1998. The current performance assessment (*Hanford Immobilized Low-Activity Waste Performance Assessment: 2001 Version* (Mann et al. 2001) was also approved. This performance assessment is also being maintained with the last summary issued in October 2002 (Mann 2002b).

Information on disposal facility design comes from information being generated in the detail design of the Integrated Disposal Facility (Comstock and Aromi 2003). Inventory for the various types of waste as well as waste form release data and methods come from the performance assessment activities mentioned above as well as from related activities. Geologic, hydrologic, and geochemistry data as well as the methods for transport simulation come from the ILAW performance activities since the IDF site was analyzed as part of the ILAW performance assessment (Mann et al. 2001).

1.5 STRUCTURE OF THIS RISK ASSESSMENT

The structure of this document follows the guidance found in *Format and Content Guide for U.S. Department of Energy Low-Level Waste Disposal Facility Performance Assessments and Composite Analyses* (DOE 1999a). However, the level of detail in this document is less than found in most performance assessments. The major chapters of this document are:

1. Introduction
2. Disposal Facility Description
3. Analysis of Performance
4. Results of Analysis
5. Results For An Inadvertent Intruder Scenario
6. Interpretation Of Results
7. Recommendations
8. Preparers, and
9. References.

2.0 DISPOSAL FACILITY DESCRIPTION

This section summarizes the relevant Hanford site characteristics, the waste characteristics, and the proposed IDF concept that is analyzed under this risk assessment. Most of the relevant Hanford Site characteristics are taken directly from the 2001 ILAW PA Data Packages (Mann and Puigh 2000) that were developed to support that performance assessment. A description of the waste characteristics is provided for the different waste forms proposed for disposal within the IDF. Finally, the relevant features of the IDF are described.

2.1 HANFORD SITE CHARACTERISTICS

The regional and local environments in which the proposed disposal activities will be located have been described in detail in the 2001 ILAW PA (Mann et al. 2001). Extensive research has been done on the physical characteristics of the Hanford site. In addition, significant data specific to the proposed IDF site have been accumulated in support of the ILAW PA. Please refer to the 2001 ILAW PA (Mann et al. 2001) and the associated data packages developed for that PA (Mann and Puigh 2000) for information relating to the geography of the Hanford Site, demographic data, past land and water use, climate and meteorology information ecology and biotic conditions, regional geology, regional hydrology, natural resources, and regional background contamination and Hanford Site monitoring.

This section will focus on the Hanford Site characteristics important to this risk assessment. Specifically, this section will describe the location of the disposal site, future Hanford use, geology of the proposed disposal location, natural recharge rates, and geochemistry relevant to the risk assessment for the proposed disposal action.

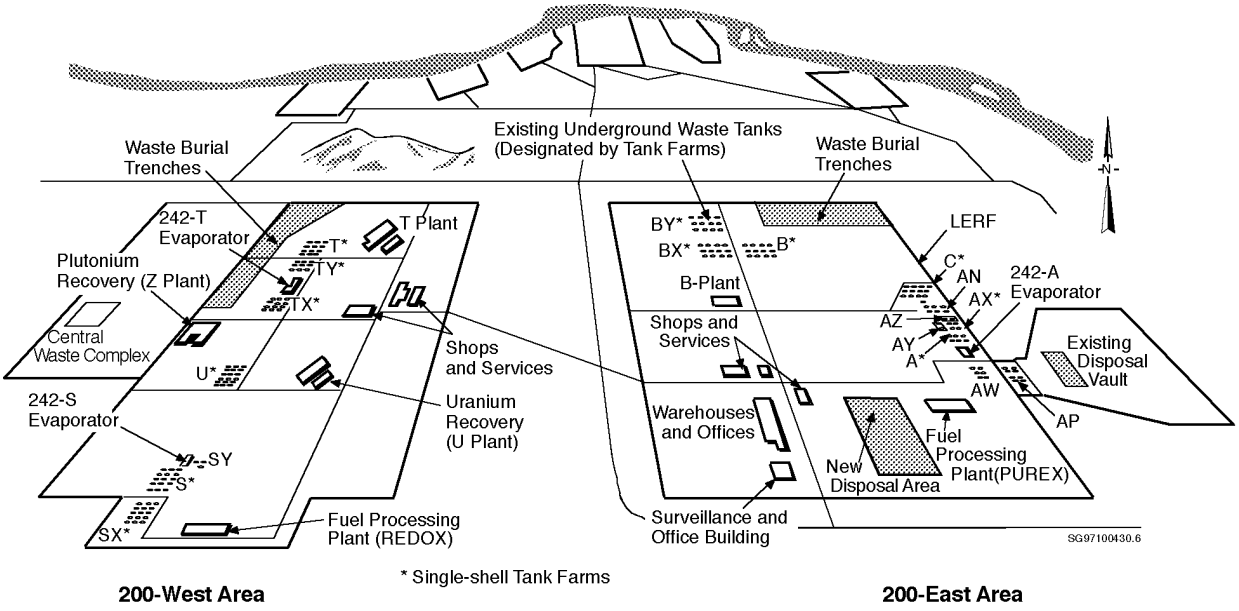
2.1.1 Location of the Disposal Site

The proposed location for the IDF is in the south-central part of the 200 East Area between existing office structures and the PUREX fuel reprocessing facility. This site is not near enough to impact any existing or past practice waste disposal sites. The location of the IDF is at the ILAW site that was chosen (Rutherford 1997) for the following three reasons (Shord 1995):

- The location is near existing tank farms
- Unused land is available
- The location is inside the fence line of the 200 Areas.

Because space was available at the ILAW site to dispose of all of Hanford's non-CERCLA wastes, the Hanford DOE Field Managers are considering the site for the disposal of all such waste. Figure 2-1 shows the proposed location for the IDF (labeled "New Disposal Area" in the figure).

Figure 2-1. Activities in the 200 Areas. The plan area for IDF is located in the south central part of the 200 East Area and is labeled "New Disposal Area."



2.1.2 Future Hanford Use

In 1992, DOE, EPA, and Washington State Department of Ecology (Ecology) gathered a group of stakeholders to study potential future uses for the Hanford Site land. This Hanford Future Site Uses Working Group issued a summary (HFSUWG 1992a) and a detailed report (HFSUWG 1992b) of its findings. The *Final Hanford Comprehensive Land-Use Plan Environmental Impact Statement* (DOE 1999c) and *Comprehensive Land-Use Plan* (DOE 1999d) are heavily based on the work of the Hanford Future Site Uses Working Group. However, DOE's land use planning extends for only 50 years instead of the 100 years forecast by the working group.

The HFSUWG 1992a-1) stated

> *"The working group identified a single cleanup scenario for the Central Plateau. This scenario assumes that future uses of the surface, subsurface and groundwater in and immediately surrounding the 200 West and 200 East Areas would be exclusive. Surrounding the exclusive area would be a temporary surface and subsurface exclusive buffer zone composed of at least the rest of the Central Plateau. As the risks from the waste management activities decrease, it is expected that the buffer zone would shrink commensurately."*

The record of decision (DOE 1999d) for the *Final Hanford Comprehensive Land-Use Plan Environmental Impact Statement* (DOE 1999c) identifies near-term land uses for the Hanford Site. The record of decision prescribes the use in the 200 Areas as exclusively industrial (primarily waste management) with much of the surrounding land having the use of preservation or conservation. Recently, the Hanford Reach National Monument (Clinton 2000) was established along the river corridor as well in lands at the northern and western edges of the site.

Most recently, DOE, EPA, and Ecology (DOE 2002) put forth a risk framework, delineating the following land use scenarios.

"The Core Zone (200 Areas including B Pond (main pond), and S Ponds) will have an Industrial Scenario for the foreseeable future.

The Core Zone will be remediated and closed allowing for "other uses" consistent with an industrial scenario (environmental industries) that will maintain human presence in this area, which in turn will enhance the ability to maintain the institutional knowledge of wastes left in place for the future generations. Exposure scenarios used for this zone should include a reasonable maximum exposure to a worker/day user, to possible Native American users, and to intruders.

DOE will follow the required regulatory processes for groundwater remediation (including public participation) to establish the points of compliance and remedial action objectives. It is anticipated that groundwater contamination under the Core Zone will preclude beneficial use for the foreseeable future, which is at least the period of waste management and institutional controls (150 years). It is assumed that the tritium and iodine-129 plumes beyond the Control Zone Boundary will exceed the drinking water standards for the period of the next 150 to 300 years (less for the tritium plume). It is expected that other groundwater contaminants will remain below, or be restored to drinking water levels outside the Core Zone.

No drilling for water use or otherwise will be allowed in the Core Zone for the foreseeable future. An intruder scenario will be calculated in assessing the risk to human health and environment.

Waste sites outside the Core Zone but within the Central Plateau (200N, Gable Mountain Pond, B/C Crib Controlled Area) will be remediated and closed based on evaluation of multiple land use scenarios to optimize land use, institutional control cost, and long-term stewardship.

An industrial use scenario will set cleanup levels on the Central Plateau. Other scenarios (e.g., residential, recreational) may be used for comparison purposes to support decision making especially for:

The post-institutional control period (>150 years)

Sites near the Core Zone perimeter to analyze opportunities to "shrink the site".

Early (precedent-setting) closure/remediation decisions.

This framework does not deal with the tank retrieval decision."

Table 2-1 summarizes this agreement.

Table 2-1. Hanford Site Land Uses. [a]

Time (Y)	Core Zone (~200 Area)	Beyond Core Zone	National Monument and Columbia River
2000→2012	DOE cleanup activities	DOE cleanup activities	DOE cleanup activities
2012→2035	DOE cleanup activities	DOE cleanup activities	Recreational use
2035→2150	Restricted industrial use; no intruders and no groundwater use	Restricted Use, no groundwater use	Recreational use
2150→X[b]	Industrial use; data for informational use only	Multiple land use, data for informational use only	Recreational use
X[b] →	Industrial use; other uses for informational use only	Multiple land use	Recreational use

a Attachment of letter of DOE 2002

b X is defined as the time that the groundwater contamination falls below the limits set in 40 CFR 141 (National Primary Drinking Water Standards) for a particular location due to contamination release before the year 2000 from Hanford Site facilities. Thus, it is likely that for locations beyond the core zone, X will be nearer to the present than for locations in the Core Zone. It is assumed (in the reference cited) that X is larger than 2150.

2.1.3 Geology of the Proposed Disposal Location

The stratigraphy at the IDF site consists of the Hanford formation and the Ringold Formation overlying the Columbia River Basin Group. Surfacial sediments are mainly Eolian deposits consisting of reworked Hanford formation sands and silts.

The stratigraphy model developed for this risk assessment is based on borehole well logging and specific characterization boreholes performed for the ILAW disposal action. This information is provided in detail in Reidel and Horton (1999) and is summarized in the 2001 ILAW PA (Mann et al. 2001). Figure 2-2 provides a summary diagram of a west to east cross-section for the IDF site.

2.1.4 Natural Recharge Rates

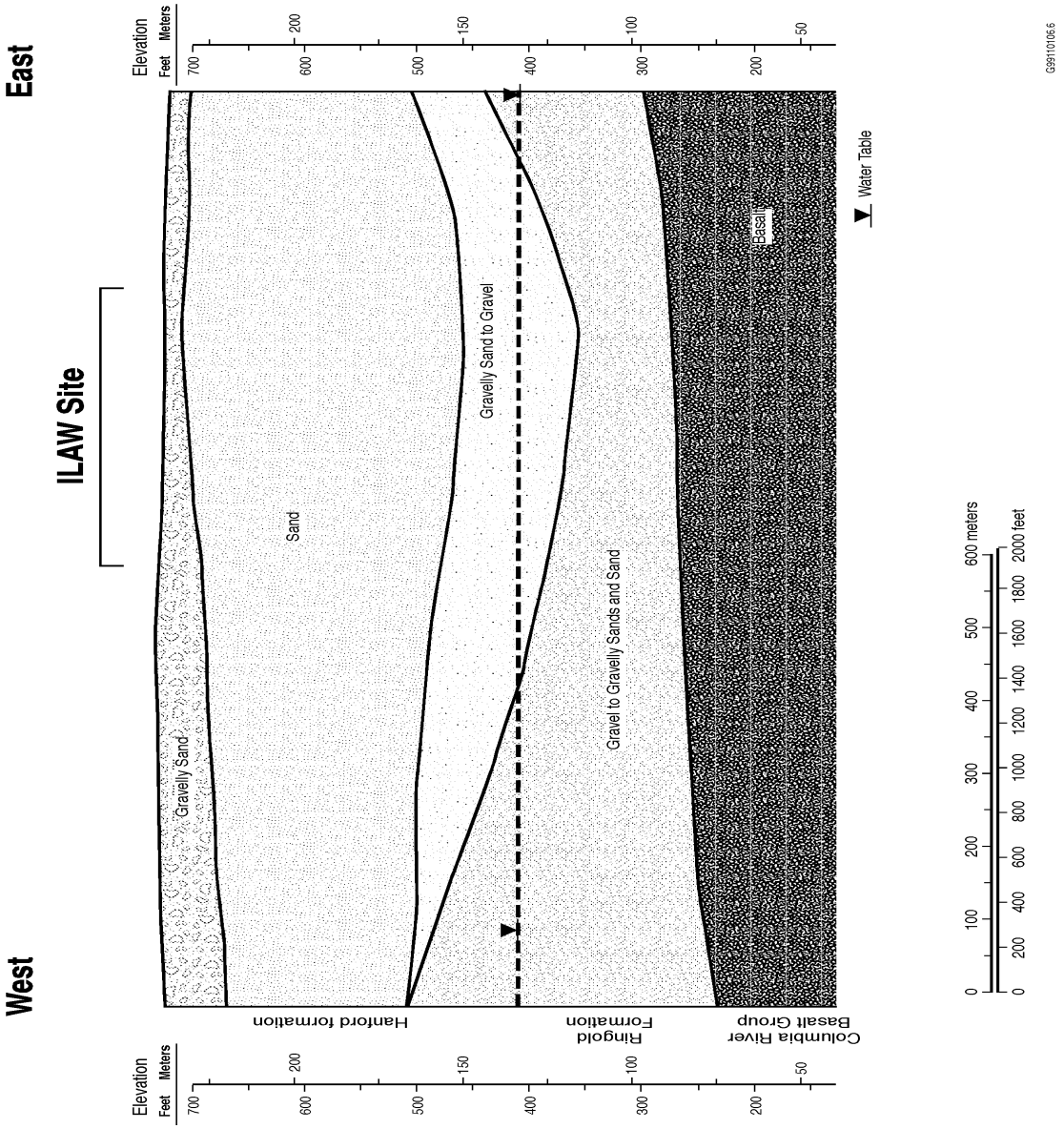
The information in this section is based on *Recharge Data Package for the Immobilized Low-Activity Waste 2001 Performance Assessment* (Fayer 1999). Recharge is the amount of total precipitation that infiltrates into the unsaturated zone (vadose zone) after runoff, evaporation, and transpiration by plants have occurred. Recharge from rain and snowmelt is a major hydrologic variable affecting contaminant transport from the proposed Integrated Disposal Facility.

The recharge rate depends on the seasonal distribution of precipitation, type of surface soil and vegetation, and climatic conditions. Maximum recharge events occur following the wettest winter periods. Under normal conditions, the recharge rate is highest in coarse-textured soils without vegetation and is at the measurement threshold in fine-textured soil with or without

vegetation. Coarse soil surfaces that are either vegetated with shallow-rooted species or bare exhibit recharge on the order of 50 percent of the precipitation.

For the IDF disposal site, surface soils are dominated by Rupert Sands and Burbank Loamy Sand. Fayer (1999) estimates that the natural recharge rates through the two types of soils are 0.9 mm/y and 4.2 mm/y, respectively. See Section 3.3.6 for a more complete description of recharge rates and the choice of values used in this risk assessment.

Figure 2-2. Summary Diagram of a West to East Cross-Section for the Proposed IDF Disposal Site (indicated as the ILAW Site in the figure).



2.2 WASTE CHARACTERISTICS

For this risk assessment we have assumed the following waste streams to be disposed in the IDF:

- ILAW
- LLW/MLLW
- Failed or decommissioned melters from the Waste Treatment and Immobilization Plant (WTP)

The source of ILAW will be the vitrification of the low-activity fraction of the Hanford tank waste after suitable separations. The source of low-level waste and mixed low-level waste will be from both onsite operations and remediation and offsite shipments from other DOE sites. The source of melters will be from the operation of the Waste Treatment Plant (WTP).

Not considered in this risk assessment is the disposal of any other waste streams. Specifically, not considered are any alternate Hanford tank waste forms such as steam reformed, bulk vitrification, or grouted Hanford tank waste.

2.2.1 Immobilized Low-Activity Waste

The TWRS record of decision (DOE 1997) states that the waste will be retrieved from the tanks, and then chemically separated to form the high-level and low-activity radioactive waste fractions. The high-level radioactive waste fraction will contain most of the radionuclides. This waste fraction will be vitrified, and the product stored until it can be transferred to a licensed high-level waste repository. The low-activity radioactive waste fraction contains the bulk of the nonradioactive chemicals and is predominantly the soluble components of the tank waste. This waste fraction will be solidified in a glass or other form that meets the DOE specifications.

It is proposed to dispose of the immobilized low-activity waste form onsite in a manner that allows the waste to be retrievable for at least 50 years, although this time period has not been adopted officially.

After waste-type separation, the low-activity waste will be immobilized into glass. Current plans involve vitrification in a joule-heated ceramic DuraMelter. The DuraMelter vitrification system imposes certain operational and process requirements on the glass formulations that include:

- Viscosity limits of 1 to 15 Pa·s at 1100°C
- Electrical conductivity limits of 0.2 to 0.7 S/cm at 1100 to 1200°C
- Liquidus temperature below 950°C.

Other factors affecting melter operations that are also important include:

- Ability to retain sulfur in the glass matrix without the formation of molten salt phases during processing; these phases are more corrosive, electrically conductive, and fluid than the glass melt, and have lower melting points.

- Compatibility of the glass melts with the projected glass contact refractory (primarily Monofrax K-3) and the metallic components of the melter (e.g., electrodes, bubblers, thermowells, etc.).
- In addition to these processing constraints, the DOE imposes additional product acceptance constraints. Detailed specifications regarding waste package size, compressive strength, crystallinity, etc., have been developed (DOE/ORP 2000).

A large number of LAW glasses have been formulated by staff at the Vitreous State Laboratory (VSL) in Washington D.C. that meet these processing and product acceptance requirements while achieving waste loadings ranging from 6 to 31 mass%. Supplemental to the VSL work, a set of 77 glasses was formulated and tested under a project funded by DOE headquarters (EM-50) (Vienna et al. 2000). The combined set of these glasses covers a very wide-ranging, multidimensional compositional space.

The LAWABP1 glass was used to represent the ILAW glass performance in the 2001 ILAW PA (Mann et al. 2001). This risk assessment will also utilize the performance of LAWABP1 glass to represent the performance of ILAW in the proposed IDF disposal action. Although LAWABP1 glass composition is not the same as the glass compositions selected by the WTP contractor, it is chosen to represent the glass composition because it was the glass composition analyzed in the base analysis of the 2001 ILAW PA and because it is now known that LAWABP1 glasses perform slightly more poorly than the WTP selected glass. See the 2001 ILAW PA (Mann et al. 2001), the 2002 ILAW PA annual summary (Mann 2002b), and McGrail et al. (2001) for additional details on the performance of LAWABP1 glass waste form.

The physical, chemical, and radiological properties of the waste at the time of disposal have not been completely determined. At the time of the start of this analysis, the ILAW waste form is expected to be contained in the form of right circular, steel cylinder containers (1.22 m diameter by 2.29 m tall) based on modification 12 of the BNFL contract (see DOE/BNFL 1998). The containers are assumed to be filled to 85% by volume waste glass. Given the geometry of the ILAW glass container, the corresponding glass height in the container is estimated to be 2.0 m.

Based on Case 3 of the TWRSO&UP (Kirkbride 1999), 158,105 m^3 of ILAW glass will be produced for Phase 1 and Phase 2 operations of the WTP. Also, 15,021 m^3 of HLW glass will be produced. The ILAW glass density is approximately 2.6 kg/L. Table 2-2 summarizes the ILAW volume, mass, and number of packages for the reference case (Case 3 of the TWRSO&UP) (Kirkbride 1999).

Table 2-2. Summary of ILAW and HLW Glass Production.

	Total MT glass	Total m^3 glass
ILAW	411,073	158,105
HLW	39,055	15,021

Each ILAW package contains 2.3 (=3.14*2*[1.22/2]^2) m^3 of ILAW glass. The total number of ILAW packages is estimated to be 70,064 for Phase 1 and Phase 2 combined. An upper bound estimate for the number of ILAW waste packages assumes ~ 15% more glass is produced. This results in approximately 81,000 ILAW waste packages being used.

2.2.2 Low-Level Waste /Mixed Low Level Waste

The baseline inventory and associated volume estimates come from the same source, the 2002 SWIFT forecast. In this forecast for waste to be disposed in 2007 and beyond, the LLW volume is 111,000 m^3 and the MLLW volume is 53,500 m^3, for a grand total of 164,500 m^3. The LLW volume estimate does not include the small amount of waste projected for disposal between 2003 and 2006. The upper bound inventory and volume estimates are taken from the SWPEIS (DOE/EIS-0286D 2003). The inventory estimate is based primarily on average concentrations of radionuclides in disposed and stored mixed and low level waste at Hanford from 1996 through 1997. To generate inventory estimates, these concentrations were multiplied by volume estimates based on forecasted waste volumes. The volumes were derived from the 1999 SWIFT forecast plus the DOE complex PEIS evaluation that attempts to encompass offsite generators who currently do not ship to Hanford but may if Hanford becomes a complex-wide disposal facility for LLW and MLLW. The upper bound volume associated with the upper bound inventory (see Table 3-2 in Section 3.1.2) is 306,000 m^3 for LLW and 188,000 m^3 for MLLW, a grand total of 494,000 m^3.

2.2.3 Waste Treatment Plant Failed Melters

The number of melters requiring disposal for the balance of the WTP mission is estimated to be 46 melters, including 18 HLW melters, 22 LAW melters, and 6 contingency melters (Zuberi and Lowe 2003). The HLW and LAW melter radiological and physical characteristics have not been finalized. The following assumptions have been made concerning the quantity of waste material remaining in the failed melters, their dimensions and overpack design.

From Duratek HLW melter drawings (WTP-M-21100, Rev. 1 sheets 1-6) the size of the original melt tank cavity is 2.44 m L x 1.52 m W x 1.17 m H (96 inches L x 60 inches W x 46 inches H). With brick corrosion and soak-in glass, the tank size grows horizontally by 35.6 cm (14 inches) (because both side walls recede) and vertically by 5 inches. The maximum final glass tank size is therefore 2.79 m L x 1.88 m W x 1.30 m H (110 inches x 74 inches x 51 inches). Upon cooling only the glass height is assumed to change from 1.30 m (51 inches) to 1.09 m (43 inches). Therefore the final maximum glass volume remaining in a HLW melter is (5,753 L) (351,378 $inches^3$) or 1.50×10^4 kg. The glass density (cold) is 2.6 kg/L.

From Duratek LAW melter drawings the size of the original melt tank cavity is 4.93 m L x 2.03 m W x 0.76 m H (194 inches L x 80 inches W x 30 inches H). With brick corrosion and soak-in glass, the tank size grows horizontally and vertically by 35.6 cm (14 inches). The final glass LAW tank size is therefore 5.64 m L x 2.74 m W x 1.12 m H (222 inches x 108 inches x 44 inches). Assuming the hot density of the LAW glass is 2.2 kg/L, the LAW cold glass density is 2.6 kg/L, and all the shrinkage occurs in the height dimension, then the final LAW glass height is 0.97 m (37 (=44*2.2/2.6) inches). Therefore, the final maximum glass volume remaining in a LAW melter is 14,524 L (887,112 $inches^3$) or 3.78×10^4 kg.

The melter Overpacks are currently designed as carbon steel containers that provide the necessary shielding, contamination control, and structural rigidity to allow direct burial of the spent/failed melters as MLLW (Zuberi and Lowe 2003). Table 2-3 summarizes the current melter overpack design envelopes.

Table 2-3. Melter and Melter Overpack Characteristics.

Melter Type	Height [1]	Length [1]	Width [1]	Surface to Volume Ratio	Melter Weight [2,3]	Overpack Weight [3]
HLW	4.38-m (172-in)	5.29-m (208-in)	5.29-m (208-in)	1.215 m^{-1}	100.8 MT (222,200 lbs)	226.8 MT (500,000 lbs)
LAW	4.86-m (190-in)	6.79-m (262-in)	9.38-m (367-in)	0.9295 m^{-1}	329.3 (MT) (726,000 lbs)	Under development [4]

1 The LAW and HLW melter dimensions include the melter overpack.

2 Melter weight includes melter filled with glass.

3 All Weights in table include a 10% contingency due to status of design, as design progresses these values will be revised

4 The recommended Land Disposal Restriction (LDR) compliance strategy for the LAW melter is to transport the melter to the disposal trench and treat the melter in-trench (encapsulate melter in grout), however, it is recognized that this path forward may include regulatory hurdles and life-cycle cost impacts greater than originally anticipated. Therefore, a backup approach was recommended by BNI that includes macro-encapsulation of the LAW melter in a sealed stainless steel box. This would be the limiting case relative to transportation limits (e.g., road load bearing limits).

Current planning has each HLW melter encapsulated in an 8-inch thick carbon steel rectangular overpack. Planning for the disposal of the LAW melters is still under development. For this risk assessment, we have assumed that a 1-inch thick steel rectangular overpack is used for the LAW melters. For this risk assessment both the LAW and HLW melters are assumed to be grouted into their respective overpacks.

2.3 DISPOSAL FACILITY DESIGN

The IDF trench will be constructed on the ILAW disposal site. Figure 2-3 shows the potential location of trench within the disposal site. The design parameters used in this risk assessment is based on the 30 % design review information for the IDF. The IDF trench conceptual model based on the 30% design review information (Comstock and Aromi 2003) is depicted in Figure 2-4. The IDF trench internal dimensions, based on the inner liner, are 375 m wide at the bottom by 13.2 m deep. The length of the trench will be sized to accommodate the waste added to the IDF. One estimate to accommodate all ILAW, LLW/MLLW (including melters) and alternate concept LAW waste would require the IDF trench to be 400 m long. The trench sides have a 3:1 slope. The bottom of the trench has a 1% downward slope from south to north to facilitate the collection of leachate at the north end of the facility (see Figure 2-5).

The trench is provided with a RCRA compliant primary and secondary liner as depicted in Figure 2-6. Beneath both the primary and secondary liner is a 0.9-m admix layer. The prepared subgrade material beneath the admix liner is assumed to be composed of backfill material. The 0.9 m operations layer is assumed to be backfill material. Because the liners have relatively short design lives (at most hundreds of years), the liners are not considered in the simulations.

The leachate collection and recovery system (LCRS) drainage gravel provides the drainage path for the two drainage (leachate) collection systems associated with this

RCRA-compliant disposal facility (see Figure 2-5). Both the primary and secondary drainage layers consist of a geocomposite drainage layer on top of high-density polyethylene (HDPE).

Because the trench walls have a fairly shallow slope (3 m run for every 1 m rise) each successive vertical layer of waste can be increased in both length and width.

The IDF trench is assumed to include backfilled soil around and on top of the waste containers in the facility. The soil has been included in this concept for the following three reasons:

- For structural support
- To wick moisture away from the waste containers
- To provide radiation shielding for the facility workers.

Above the IDF trench is a surface barrier designed to minimize physical intrusion risk and recharge. This surface barrier has not yet been designed for the IDF. Beneath the surface barrier, a sand-gravel capillary break is assumed that will divert any moisture that may come through the surface barrier away from the trench. These two barriers implement the goal of minimizing the amount of water that enters the trench. The barrier is assumed to be high enough to ensure the depth of the waste packages within the trench is at least 5 meters from the top surface of the barrier. This minimum depth is required by NRC rules (10 CFR 61) for the ILAW waste form. The extent of the barrier beyond the inner dimension of the trench is assumed to be 10 meters (Burbank 2002). The current preconceptual design has a modified RCRA-compliant subtitle C barrier with a 2 percent slope.

The current operational plans for the IDF are to fill the trench in stages. For this risk assessment we have assumed the ILAW and MLLW (including melters) would be placed into one cell (1/2 trench width in the East – West direction) and the LLW would be placed into the other cell (see Figure 2-6). The length of the trench southward would be extended, as necessary, to accommodate the near term waste inventories planned for disposal. The waste package loading for the different waste forms is assumed to consume 40% of the available trench volume within the IDF. See Section 3.3.6 for an estimate of the fraction of the IDF occupied by each waste form. For the purposes of this risk assessment the facility closure is assumed to occur in 2046. This date is later than used in the 2001 ILAW PA and was chosen to accommodate the disposal of LLW and MLLW at the IDF site. This date impacts the intruder and air pathway dose estimates for only the short half-life radionuclides.

Figure 2-3. Layout of Integrated Disposal Facility.

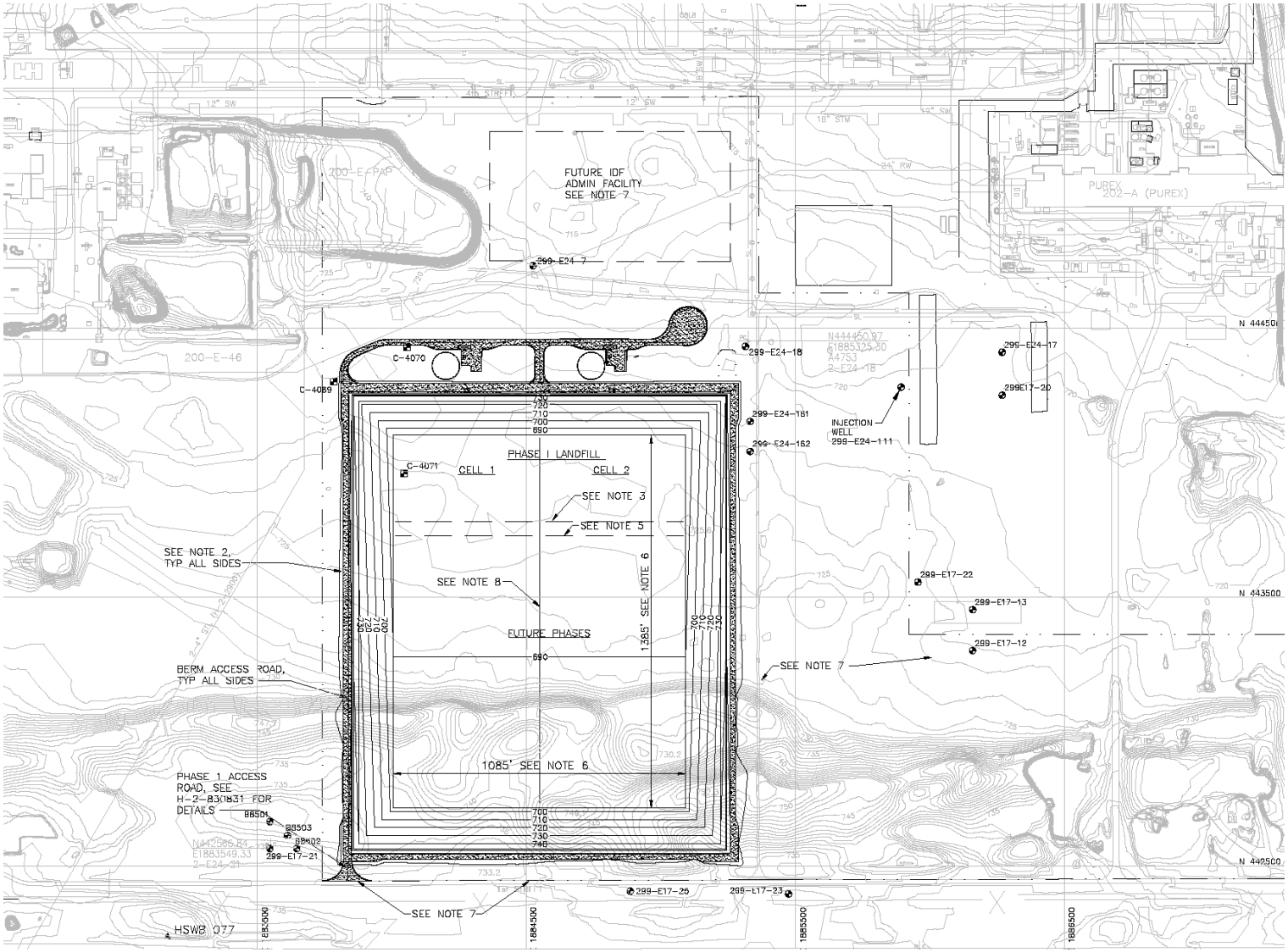
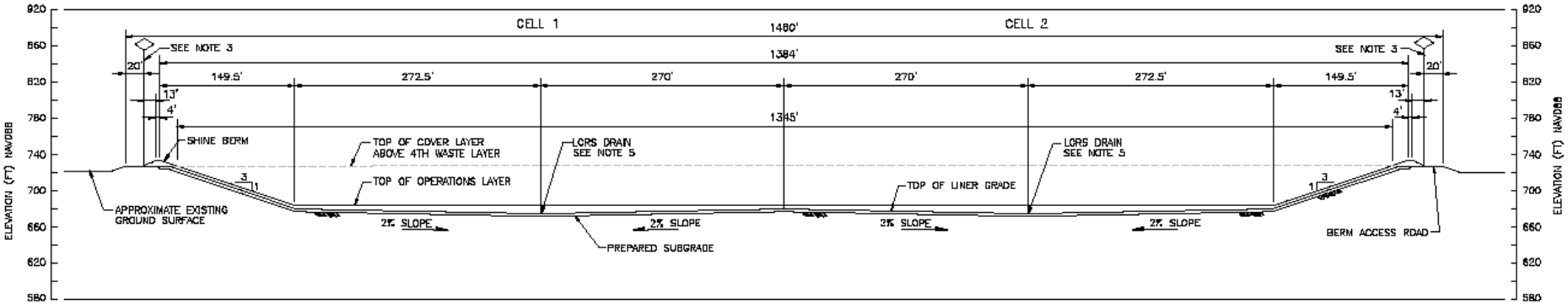


Figure 2-4. IDF Trench Conceptual Model.

East – West Cross-Section



North – South Cross Section

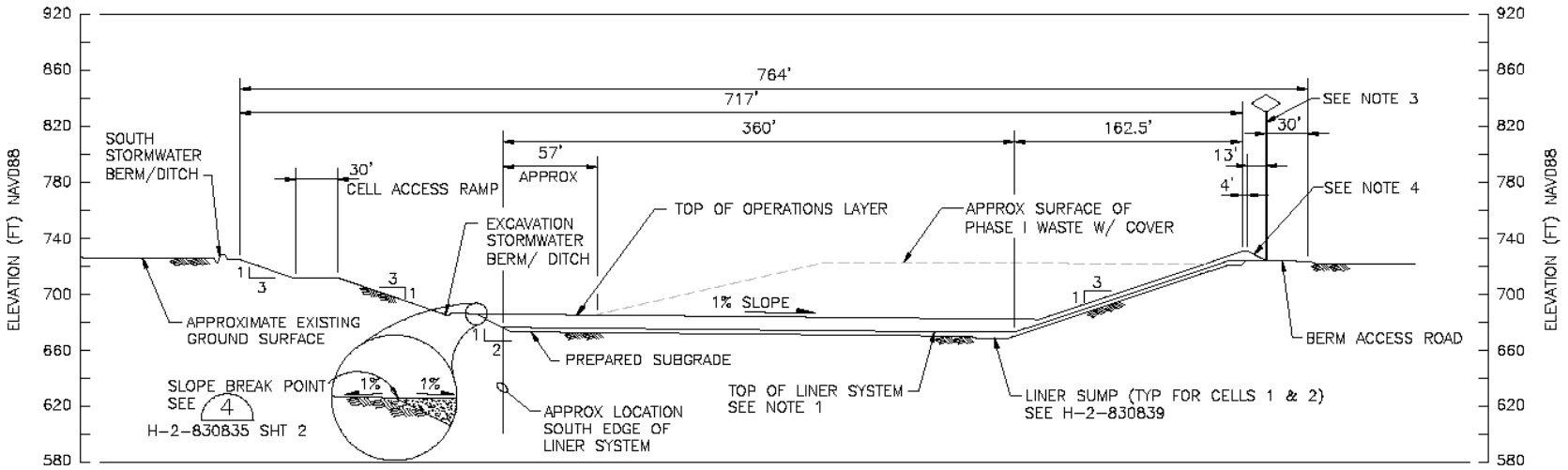


Figure 2-5. IDF Leachate Collection and Recovery System Location.

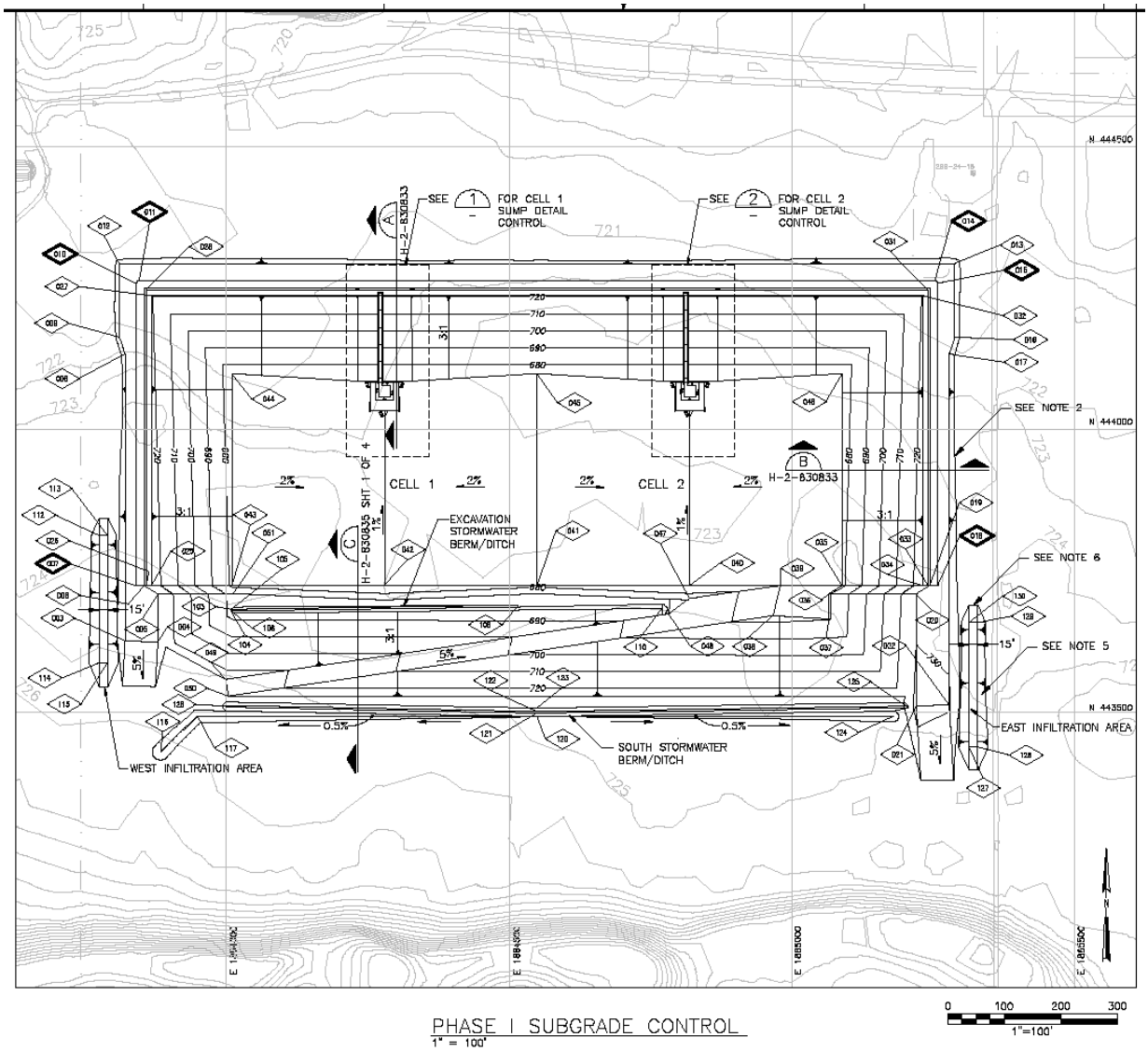
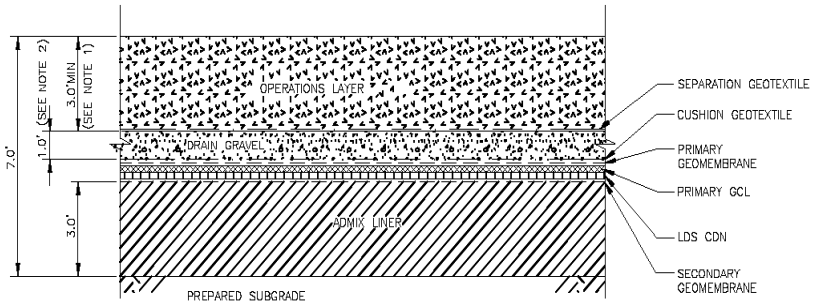


Figure 2-6. IDF RCRA Compliant Liner Details.



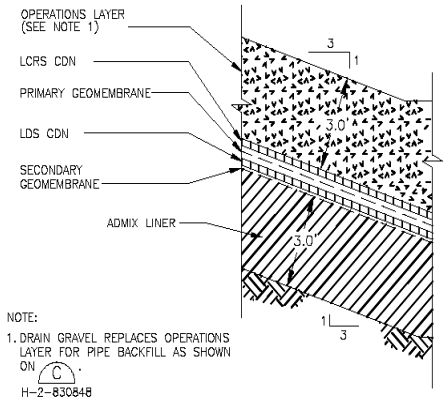
NOTES:

1. OPERATIONS LAYER THICKNESS VARIES ACROSS CELL BOTTOM WITH A 3-FOOT MIN. THICKNESS.
2. INCREASE DRAIN GRAVEL THICKNESS IN VICINITY OF LEACHATE COLLECTION AND RISER PIPES IN LCRS SUMP AS SHOWN ON (B) AND (D) H-2-830848 H-2-830845

BOTTOM LINER DETAIL (1)

NTS

H-2-830836, H-2-830839 H-2-830840



GENERAL NOTE (H2-830838 THROUGH 830840)

SPACING BETWEEN GEOSYNTHETIC LAYERS IS EXAGGERATED FOR CLARITY.

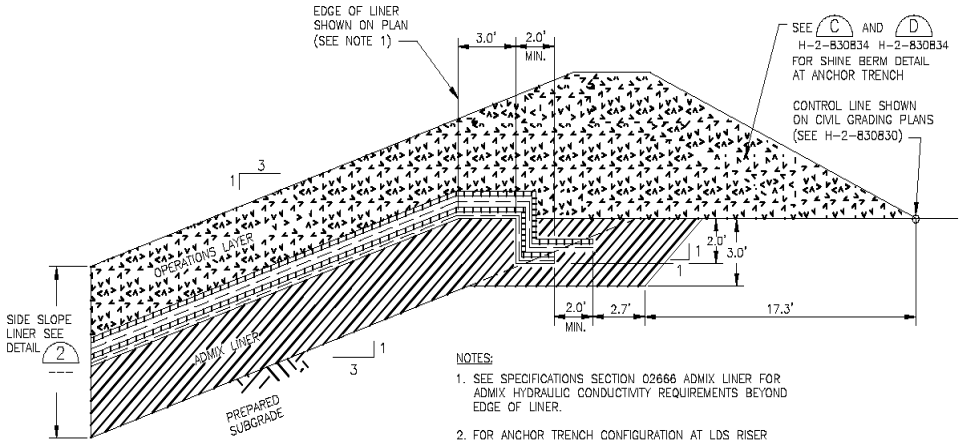
NOTE:

1. DRAIN GRAVEL REPLACES OPERATIONS LAYER FOR PIPE BACKFILL AS SHOWN ON (C). H-2-830848

SIDE SLOPE LINER DETAIL (2)

NTS

H-2-830836, H-2-830839, H-2-830840, H-2-830848



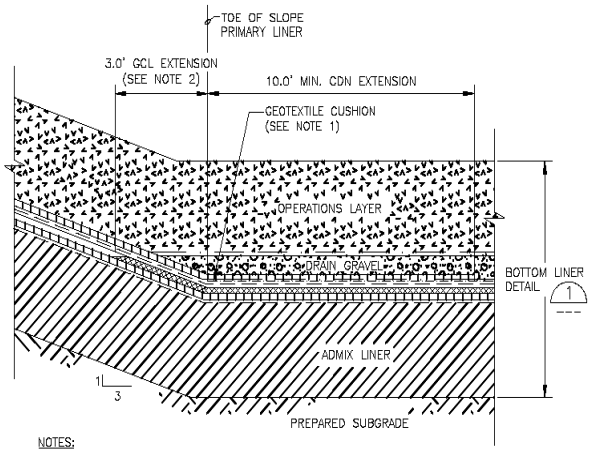
NOTES:

1. SEE SPECIFICATIONS SECTION 02666 ADMIX LINER FOR ADMIX HYDRAULIC CONDUCTIVITY REQUIREMENTS BEYOND EDGE OF LINER.
2. FOR ANCHOR TRENCH CONFIGURATION AT LDS RISER PIPE TRENCH SEE SECTION C ON DWG. H-2-830847.

LINER ANCHOR TRENCH DETAIL (3)

NTS

H-2-830834, H-2-830836



NOTES:

1. GEOTEXTILE CUSHION ENDS AT TOE OF SLOPE
2. EXTEND GCL 3.0' UP SLOPE (HORIZONTAL LENGTH) TO TOP OF DRAIN GRAVEL.

TOE OF SLOPE LINER DETAIL (4)

NTS

H-2-830836, H-2-830839

3.0 ANALYSIS OF PERFORMANCE

This chapter describes the models and input data used to analyze the long-term environmental performance of the IDF. For the analyses, the information discussed in Chapter 2 is translated into a conceptual physical model, then into a numerical model. The chapter also provides justification for these translations.

The strategy for this risk assessment was to define and analyze a reference case and other cases that provide insight into the disposal system's probable performance. The reference case was developed using best information for the environmental, waste form, and disposal facility parameters and how the parameters will change with time. These best estimates for the Hanford Site and the ILAW waste disposal are defined and justified in separate published reports that have been combined in *Data Packages for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment: 2001 Version* (Mann and Puigh 2000). The best estimates for the LLW and MLLW contaminant release models were taken from the Solid Waste Burial Ground (SWBG) PAs (Wood et al. 1995a and 1996).

This chapter shows how the physical systems presented in Chapter 2 are translated into the numerical models that produce the results presented in Chapters 4 and 5. The chapter covers the following topics:

Inventory Source (Section 3.1) - Describes the radionuclide inventories for ILAW, LLW/MLLW and the melters.

Pathways and Scenarios (Section 3.2) - Explains the pathways and scenarios that were analyzed.

Values and Assumptions (Section 3.3) - Presents the assumptions use in the analyses, including the actual data.

Performance Assessment Methodology (Section 3.4) - Presents methodology used in the analyses, including the actual data used.

3.1 INVENTORY SOURCE

Inventory estimates are needed to assess the estimated impacts of the proposed disposal action against performance objectives established to protect human health and the environment (see Table 1-1). The 2001 ILAW PA inventory is assumed for the ILAW waste form. A preliminary estimate for the LLW/MLLW inventory is taken from the most recent Solid Waste Integrated Forecast Technical (SWIFT) report (Barcot 2002) and bounding estimates developed for the Hanford Site SWPEIS (DOE/EIS-0286D 2003). The inventory contained in the disposed melters is taken from information provided for the tank inventory and ILAW inventory used for the 2001 ILAW PA.

3.1.1 Immobilized Low-Activity Waste

The ILAW inventory assumed in this risk assessment is provided by Wootan (1999) and is equivalent to the inventory used in the 2001 ILAW PA. This inventory is provided in Table 3-1. For the reference case in this document, we have assumed that all the technetium in the Tank Waste inventory has been incorporated into the ILAW waste.

Table 3-1 provides the total inventory in the tanks and in the ILAW packages, as well as the expected average and maximum concentration in the ILAW packages. The best basis tank-by-tank inventories (BBI) as of October 1, 1998, were adjusted for waste transfers not accounted for in the BBI, and for non-BBI analytes that are in the waste treatment contract. The BBI inventories were adjusted to a common date (October 1, 1994). The BBI values are based on a tank-by-tank evaluation of measurements from a tank, as well as modeling results of transfers to and from the tank. These values are listed as the tank inventory in Table 3-1. The nominal ILAW inventories for all the materials explicitly included are based on the Tank Waste Remediation System Operation and Utilization Plan (Kirkbride 1999). The ^{99}Tc inventory in ILAW has been set equal to the tank inventory based on the decision to remove technetium ion exchange system from WTP (Schepens 2003) and assuming all the inventory goes into the low-activity waste stream. The upper bound ILAW inventory given in Table 3-1 represents the estimated upper bound for these inventories in ILAW. The upper bound radionuclide estimates are based on either contract limits (for strontium, technetium, cesium, neptunium, plutonium, americium, and curium) (DOE/BNFL 1998) or are taken to be the BBI tank inventories without separation. The only exceptions to this approach are for ^{79}Se and ^{99}Tc. The upper bound estimate for ^{79}Se is assumed to be 30% higher than the estimated tank inventory and the ^{99}Tc upper bound estimate is assumed to be 15% higher than the estimated tank inventory. The upper bound chemical estimates are either the estimated tank inventory or two times the inventory. (See Wootan [1999] for a discussion on the basis for these estimates.) The average package concentration is calculated by dividing the total inventory for each contaminant by the estimated volume (1.581×10^5 m^3 from Table 2-2). The maximum batch concentration is estimated from the comparison of the batch-to-batch variation in Kirkbride's (1999) flow process calculations to the average inventories in a waste package. These estimates reflect the tank-to-tank variation in inventory.

Table 3-1. ILAW Inventories and Concentrations for Important Constituents.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	Tank Inventory	ILAW Inventory	Upper Bound ILAW Inventory	Average Package Concentration	Maximum Batch Concentration
3-H	2.46E+04	0.00E+00	2.46E+04	0.00E+00	0.00E+00
14-C	4.38E+03	0.00E+00	4.38E+03	0.00E+00	0.00E+00
59-Ni	8.58E+02	1.67E+02	8.58E+02	1.06E-03	4.02E-03
60-Co	1.99E+04	4.18E+03	1.99E+04	2.64E-02	3.07E-01
63-Ni	8.45E+04	1.62E+04	8.45E+04	1.02E-01	3.91E-01
79-Se	5.74E+01	4.80E+01	7.45E+01	3.03E-04	5.45E-03
90-Sr+D [a,c]	5.99E+07	4.50E+06	5.85E+06	2.85E+01	5.43E+01
93-Zr	4.12E+03	1.25E+03	4.12E+03	7.94E-03	3.37E-02
93m-Nb	2.53E+03	8.36E+02	2.53E+03	5.29E-03	4.47E-02
99-Tc	2.89E+04	2.89E+04	3.33E+04	1.83E-01	4.98E-01

Table 3-1. ILAW Inventories and Concentrations for Important Constituents.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	Tank Inventory	ILAW Inventory	Upper Bound ILAW Inventory	Average Package Concentration	Maximum Batch Concentration
106-Ru+D [a]	1.27E+05	8.94E+02	1.27E+05	5.65E-03	2.59E-01
113m-Cd	1.67E+04	7.97E+03	1.67E+04	5.04E-02	2.14E-01
125-Sb+D [a]	2.47E+05	5.20E+04	2.47E+05	3.29E-01	6.50E+00
126-Sn+D [a]	4.64E+02	1.69E+02	4.64E+02	1.07E-03	4.17E-03
129-I	1.01E+02	2.20E+01	1.01E+02	1.39E-04	1.81E-03
134-Cs	8.71E+04	3.76E+02	4.89E+02	3.73E-01	1.35E+01
137-Cs+D [a,c]	6.37E+07	9.11E+05	1.18E+06	5.76E+00	7.80E+00
151-Sm	2.61E+06	7.80E+05	2.61E+06	4.93E+00	2.42E+01
152-Eu	1.45E+03	3.07E+02	1.45E+03	1.94E-03	4.21E-02
154-Eu	1.83E+05	3.77E+04	1.83E+05	2.38E-01	6.13E+00
155-Eu	1.76E+05	3.15E+04	1.76E+05	1.99E-01	7.36E+00
226-Ra+D [a,b]	6.31E-02	5.70E-02	1.14E+03	3.61E-07	1.56E-05
227-Ac+D [a,b]	8.76E+01	6.06E-02	8.75E+01	3.83E-07	1.76E-06
228-Ra+D [a,b]	7.71E+01	3.30E+01	7.75E+01	2.09E-04	1.06E-03
229-Th+D [a,b]	1.81E+00	3.40E-01	1.81E+00	2.15E-06	1.14E-05
231-Pa [b]	1.56E+02	3.44E-01	1.53E+02	2.17E-06	1.05E-05
232-Th	4.40E+00	1.28E+00	4.40E+00	8.09E-06	5.97E-05
232-U	1.49E+02	3.46E+01	1.49E+02	2.19E-04	1.64E-03
233-U	5.72E+02	1.31E+02	5.72E+02	8.26E-04	6.22E-03
234-U	3.42E+02	4.41E+01	3.42E+02	2.79E-04	1.95E-03
235-U+D [a]	1.46E+01	1.79E+00	1.46E+01	1.13E-05	7.97E-05
236-U	1.24E+01	1.43E+00	1.24E+01	9.03E-06	3.68E-05
237-Np+D [a,c]	1.85E+02	8.10E+01	3.00E+02	5.13E-04	1.78E-03
238-Pu [c]	2.70E+03	1.06E+02	3.94E+02	6.72E-04	2.69E-03
238-U+D [a]	3.28E+02	4.83E+01	3.28E+02	3.06E-04	2.02E-03
239-Pu [c]	5.55E+04	3.05E+03	1.13E+04	1.93E-02	9.50E-02
240-Pu [c]	1.13E+04	5.25E+02	1.95E+03	3.32E-03	1.34E-02
241-Am [c]	1.07E+05	1.08E+04	4.01E+04	6.85E-02	1.69E+00

Table 3-1. ILAW Inventories and Concentrations for Important Constituents.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	Tank Inventory	ILAW Inventory	Upper Bound ILAW Inventory	Average Package Concentration	Maximum Batch Concentration
241-Pu	1.66E+05	7.17E+03	1.66E+05	4.53E-02	1.98E-01
242-Cm	1.72E+02	5.76E+01	1.72E+02	3.64E-04	1.16E-02
242-Pu [c]	1.07E+00	4.49E-02	1.66E-01	2.84E-07	1.69E-06
243-Am+D [a,c]	1.76E+01	6.89E-01	2.55E+00	4.36E-06	9.01E-05
243-Cm [c]	3.47E+01	6.73E+00	2.49E+01	4.26E-05	5.18E-04
244-Cm [c]	7.84E+02	1.01E+02	3.73E+02	6.36E-04	6.77E-03
Ag^{+} (silver)	1.51E+03	1.08E+02	3.03E+03	6.83E-04	5.68E-03
As^{5+} (arsenic)	2.08E+01	1.76E+01	4.15E+01	1.12E-04	7.42E-03
Ba^{2+} (barium)	1.70E+03	1.86E+01	3.39E+03	1.17E-04	7.24E-03
Be^{2+} (beryllium)	1.09E+02	6.14E-01	2.18E+02	3.89E-06	5.48E-04
Cd^{2+} (cadmium)	4.18E+02	6.30E+01	8.36E+02	3.98E-04	5.13E-03
Cl^{-} (chlorine)	9.37E+05	9.31E+05	9.37E+05	5.89E+00	1.55E+01
CN^{-} (cyanide)	1.09E+05	0.00E+00	1.09E+05	0.00E+00	0.00E+00
Cr (TOTAL)(chromium)	6.72E+05	2.74E+05	6.72E+05	1.73E+00	1.27E+01
Cu^{2+} (copper)	3.15E+02	7.33E-01	6.31E+02	4.63E-06	2.54E-05
F^{-} (fluoride)	1.20E+06	9.94E+05	1.20E+06	6.28E+00	2.75E+01
Fe^{3+} (iron)	1.40E+06	4.48E+04	1.40E+06	2.83E-01	2.86E+00
Hg^{2+} (mercury)	2.10E+03	1.92E+02	2.10E+03	1.22E-03	3.38E-02
Mn^{4+} (manganese)	1.96E+05	1.38E+04	1.96E+05	8.71E-02	4.20E-01
NH_3 (ammonia)	5.01E+05	0.00E+00	5.01E+05	2.53E+00	4.24E+01
Ni^{2+} (nickel)	1.80E+05	3.05E+04	1.80E+05	1.93E-01	2.96E+00
NO_2^{-} (nitrite)	1.26E+07	0.00E+00	1.26E+07	0.00E+00	0.00E+00
NO_3^{-} (nitrate)	5.25E+07	0.00E+00	5.25E+07	0.00E+00	0.00E+00
Pb^{2+} (lead)	8.40E+04	7.83E+03	8.40E+04	4.95E-02	2.73E-01
Se^{6+} (selenium)	6.11E-01	5.33E-01	1.22E+00	3.37E-06	2.96E-05
SO_4^{2-} (sulfate)	3.91E+06	3.39E+06	3.91E+06	2.15E+01	9.12E+01
Tl^{3+} (thallium)	2.54E+04	NA	5.08E+04	0.00E+00	0.00E+00
Zn^{2+} (zinc)	2.89E+03	1.98E+03	5.79E+03	1.25E-02	1.19E-01

Table 3-1. ILAW Inventories and Concentrations for Important Constituents.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	Tank Inventory	ILAW Inventory	Upper Bound ILAW Inventory	Average Package Concentration	Maximum Batch Concentration
1,1,1-trichlorethane[d]	NA	0.00E+00	9.17E+02	0.00E+00	0.00E+00
1,1,2-trichloroethane[d]	NA	0.00E+00	9.17E+02	0.00E+00	0.00E+00
Benzene[d]	NA	0.00E+00	1.53E+03	0.00E+00	0.00E+00
carbon tetrachloride[d]	NA	0.00E+00	9.17e+02	0.00E+00	0.00E+00
Chloroform[d]	NA	0.00E+00	9.17E+02	0.00E+00	0.00E+00
ethyl benzene[d]	NA	0.00E+00	1.53E+03	0.00E+00	0.00E+00
methylene chloride[d]	NA	0.00E+00	4.59E+03	0.00E+00	0.00E+00
n-butyl alcohol[d]	NA	0.00E+00	3.98E+02	0.00E+00	0.00E+00
Toluene[d]	NA	0.00E+00	1.53E+03	0.00E+00	0.00E+00
trichloroethylene (1,1,2-trichloroehylene)[d]	NA	0.00E+00	9.17E+02	0.00E+00	0.00E+00
xylenes-mixed isomers (sum of m-, o-, and p-xylene)[d]	NA	0.00E+00	4.59E+03	0.00E+00	0.00E+00
1,4-dichlorobenzene[d]	NA	0.00E+00	9.17E+02	0.00E+00	0.00E+00

[a] The D indicates that the short-lived daughters of these isotopes are in equilibrium with the isotope

[b] These values have been adjusted based on the Kupfer et al. (1999) estimate for tank inventory. Inventories for radionuclides are as of 10/1/98.

[c] Upper bound ILAW inventory estimate based on contract limit

[d] Tank inventories of specific organic compounds are not available; organic compounds are not expected to survive the vitrification process. "NA" indicates components for which inventory information is not available.

3.1.2 Low-Level and Mixed Low-Level Waste

Inventory estimates for solid wastes that could be disposed in the Integrated Disposal Facility (IDF) in the 200 East Area just west of the PUREX facility are summarized in Table 3-2. Radionuclide inventory estimates from two sources (the latest annual waste forecast and input from the Solid Waste Environmental Impact Statement) are provided below.

The first source is the annual waste forecast (referred to as the Solid Waste Integrated Forecast Technical (SWIFT) Report that is derived each year by the waste management group for the currently operating Low Level Waste Burial Grounds (LLBG). In this forecast, known generators of wastes that are being disposed in the LLBG are requested to provide life cycle forecasts of future waste to be disposed in the LLBG. The inventories (in Ci decayed to 2002) provided in Table 3-2 are from the most recent SWIFT report (Barcot 2002) and are appropriate for the primary analysis (reference case).

Estimates for the average contaminant concentrations are also provided in Table 3-2. Using the volume estimates for the LLW and MLLW baseline and upper bound inventories discussed in Section 2.2.2, average contaminant concentrations in the waste were estimated by combining the LLW and MLLW inventories and dividing by the sum of the LLW and MLLW volumes for the baseline and upper bound estimates, respectively.

Radionuclide-specific activity values are provided in the SWIFT report on a year-by-year basis out to 2046 and are broken into several categories of waste. For this analysis, wastes forecasted for disposal in 2007 and beyond are assumed to be disposed in the IDF (Table 3-2). These include Category 1 Waste, Category 3 waste and Mixed Low Level Waste (MLLW). Generally speaking, the difference between Category 1 and Category 3 waste is that Category 3 waste is routinely disposed in concrete boxes or monoliths and Category 1 wastes are disposed as received, typically in 55-gallon drums and metal boxes. MLLW contains RCRA constituents in addition to radionuclides and must be treated to satisfy land disposal restrictions before disposal is permitted. For the purposes of this analysis, MLLW is assumed to be disposed like Category 3 waste.

Given the uncertainty of future waste forecasts, a bounding, higher inventory case is also recommended for select environmentally mobile radionuclides. With the exception of I-129, these values are being used to generate the Hanford Site SWPEIS (DOE/EIS-0286D 2003). These inventory estimates include wastes to be disposed beginning in 2008, a year later than the start time in the previous estimate. However, given the speculative nature of waste forecasts, this is an insignificant discrepancy relative to the base case estimates. A bounding I-129 inventory of 9 Ci is recommended on the basis of discussions with WTP contractor staff about the generation of liquid effluent wastes from Low Activity Waste (LAW) glass production during Phase I operations of the Waste Treatment Project.

A list of nonradioactive constituents is identified in the Solid Waste Information Tracking System (SWITS) for currently stored and disposed MLLW at the Hanford Site. Also, current generators for the SWIFT report have provided a very limited description of constituents in future MLLW. Of these two sources, the current waste record is the most complete. However, this record only quantifies the mass of individual constituents indirectly and may or may not be a good indicator of future waste characteristics. Nitrate and chromium inventory and concentration estimates were derived from waste properties records of stored and disposed MLLW currently at the Hanford Site that are provided in SWITS. An average concentration was determined from the ratio of cumulative inventory to total MLLW. One very large existing waste stream, 183-H basin waste, was excluded from this calculation because it is unique and makes up the majority of the stored waste volume, thereby skewing the randomness of the averaging process. To estimate a future inventory, the average concentration was multiplied by the projected MLLW volumes.

Given these limitations, it appears that MLLW will contain a wide variety of metals, inorganic and organic compounds. Crude mass estimates of individual species from current record data (e.g., the product of container volumes and weight percents for specific constituents found in the record and an assumed average density) indicate the more prevalent constituents

Table 3-2. Solid (LLW and MLLW) Waste Inventory and Concentration Estimates.

(Radionuclide inventory estimates in Ci decayed to January 2002; and concentrations in Ci/m^3)

Radionuclides	Baseline Inventory Estimates [a]						Bounding Estimates [b]					
	Low-Level Waste		Mixed Low-Level Waste [c]	Total Inventory		Concentration [f]	Low-Level Waste		Mixed Low-Level Waste [c]	Total Inventory		Concentration [g]
	Category 1	Category 3		Ungrouted [d]	Grouted [c,e]		Category 1	Category 3		Ungrouted [d]	Grouted [c,e]	
C-14	1.00E+00	1.00E+00	1.00E+00	1.00E+00	2.00E+00	1.82E-05	1.60E+01	1.50E+02	5.70E+00	1.60E+01	1.56E+02	3.48E-04
Tc-99	1.00E+00	9.00E+01	1.90E+01	1.00E+00	1.09E+02	6.69E-04	1.30E+00	3.20E+03	3.40E+02	1.30E+00	3.54E+03	7.17E-03
I-129	1.00E+00	1.00E+00	5.00E+00	1.00E+00	6.00E+00	4.26E-05	3.70E-03	9.00E+00[f]	1.10E-01	3.70E-03[m]	9.11E+00	1.84E-05
Uranium [i]	1.00E+00	2.00E+00	1.00E+00	1.00E+00	3.00E+00	2.43E-05	3.60E+00	8.60E+02	7.00E+02	3.60E+00	1.56E+03	3.17E-03
U-234 [i]	4.90E-01	9.80E-01	4.90E-01	4.90E-01	1.47E+00	1.19E-05	1.76E+00	4.21E+02	3.43E+02	1.76E+00	7.64E+02	1.55E-03
U-235 [i]	2.27E-02	4.54E-02	2.27E-02	2.27E-02	6.82E-02	5.52E-07	8.18E-02	1.95E+01	1.59E+01	8.18E-02	3.54E+01	7.19E-05
U-238 [i]	4.87E-01	9.74E-01	4.87E-01	4.87E-01	1.46E+00	1.18E-05	1.75E+00	4.19E+02	3.41E+02	1.75E+00	7.60E+02	1.54E-03
H-3	2.20E+05	1.00E+00	4.70E+02	2.20E+05	4.71E+02	1.34E+00						
Co-60	3.70E+04	6.60E+04	2.70E+01	3.70E+04	6.60E+04	6.26E-01						
Se-79	1.00E+00	1.00E+00	0.00E+00	1.00E+00	1.00E+00	1.22E-05						
Sr-90	1.10E+02	3.20E+02	3.90E+03	1.10E+02	4.22E+03	2.63E-02						
Cs-137	4.20E+01	1.80E+02	1.90E+04	4.20E+01	1.92E+04	1.17E-01						
Np-237	1.00E+00	0.00E+00	4.00E+00	1.00E+00	4.00E+00	3.04E-05						
Ra-226	1.00E+00	1.00E+00	1.00E+00	1.00E+00	2.00E+00	1.82E-05						
Pu-241	2.20E+01	2.60E+01	5.50E+04	2.20E+01	5.50E+04	3.35E-01						
Plutonium [j]	2.40E+01	3.80E+01	1.00E+00	2.40E+01	3.90E+01	3.83E-04						
Americium [k]	3.00E+00	1.00E+00	3.20E+01	3.00E+00	3.30E+01	2.19E-04						
NO^3 (nitrate) [l]	---	---	---	0	1.02E+06	1.66E+01						
Cr (TOTAL)(chromium) [l]	---	---	---	0	3.08E+04	5.0E-01						

[a] Inventory estimates provided in the 200 SWIFT report (Barcot 2002)

[b] Inventory estimates from the SWPEIS (DOE/EIS-0286D)

[c] For analytical purposes all MLLW is assumed to be grouted

[d] Ungrouted is synonymous with Category 1 disposal conditions
[e] Grouted is synonymous with Category 3 disposal conditions
[f] Concentration estimate based on total inventory divided by total volume estimate (163,500 m^3 [LLW = 111,000 m^3 ; MLLW = 53,500 m^3])
[g] Concentration estimate based on total inventory divided by total volume estimates (494,000 m^3 [LLW = 306,000 m^3 ; MLLW = 188,000 m^3])
[h] Inventory estimate based on discussions with WTP staff on estimated I-129 inventory in secondary liquid effluent waste stream from Phase 1 glass production
[i] Inventory estimate includes U-233, U-234, U-235, and U-238. For risk assessment assume natural uranium isotopic partitioning
[j] Inventory estimate includes Pu-238, Pu-239, Pu-240, and Pu-242; for risk assessment assume all inventory is Pu-239
[k] Inventory estimate includes Am-241 and Am-243; for risk assessment assume all inventory is Am-243
[l] Inventory estimated in kg, concentration in kg/m^3; concentration estimate based on assumed waste volume estimate of 61,500 m^3
[m] Bounding estimate for I-129 increased to 1.0 Ci in the report calculations to be consistent with the Baseline Inventory Estimate for I-129.

among metals to be nickel, mercury, chromium and cadmium, among inorganic compounds to be sodium nitrate, sodium hydroxide and sodium sulfate, and among organic compounds to be polycholorinated biphenyls, tetrachloroethylene, xylene, 1,1,1-trichloroethane and carbon tetrachloride. The final inventory of disposed nonradioactive constituents will also be influenced by treatment required to satisfy land disposal restrictions. In particular, organic inventories will be largely destroyed.

3.1.3 WTP Melters

The inventories associated with the HLW and LAW melters scheduled for disposal in the IDF are not known. For this risk assessment we have assumed that each melter contains the maximum amount of waste. Specifically, each HLW melter contains 1.5×10^4 kg of HLW and each LAW melter contains 3.8×10^4 kg of LAW (see Section 2.2.4).

For the LAW melters we have assumed the contaminant concentrations in the waste material remaining in the melter are the same as the ILAW glass contaminant concentrations provided in Table 3-1. Note that this relatively small amount of waste is double counted as it is also placed in the ILAW containers.

For the HLW melters we have assumed the inventory remaining in the melter is based on the tank and ILAW inventories provided in Table 3-1 and the total HLW glass volume produced (given in Table 2-2). The total HLW inventory is estimated by subtracting the ILAW inventory from the Tank inventories provided in Table 3-1 for each contaminant. These HLW inventories are conservative because they neglect any losses of contaminants to secondary waste streams. The HLW contaminant concentration is then estimated by dividing the HLW inventory for each contaminant by the total amount of HLW glass produced (15,021 m^3). Table 3-3 summarizes the anticipated inventory and contaminant concentrations in the HLW melters.

Table 3-3. Estimated HLW Glass Contaminant Concentration.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	Tank Inventory	ILAW Inventory	HLW Inventory [1]	HLW Contaminant Concentration [2]
3-H	2.46E+04	0.00E+00	2.46E+04	1.64E+00
14-C	4.38E+03	0.00E+00	4.38E+03	2.92E-01
59-Ni	8.58E+02	1.67E+02	6.91E+02	4.60E-02
60-Co	1.99E+04	4.18E+03	1.57E+04	1.05E+00
63-Ni	8.45E+04	1.62E+04	6.83E+04	4.55E+00
79-Se	5.74E+01	4.80E+01	9.40E+00	6.26E-04
90-Sr+D [a,c]	5.99E+07	4.50E+06	5.54E+07	3.69E+03
93-Zr	4.12E+03	1.25E+03	2.87E+03	1.91E-01
93m-Nb	2.53E+03	8.36E+02	1.69E+03	1.13E-01

Table 3-3. Estimated HLW Glass Contaminant Concentration.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	Tank Inventory	ILAW Inventory	HLW Inventory [1]	HLW Contaminant Concentration [2]
99-Tc	2.89E+04	2.89E+04	0.00E+00	0.00E+00
106-Ru+D [a]	1.27E+05	8.94E+02	1.26E+05	8.40E+00
113m-Cd	1.67E+04	7.97E+03	8.73E+03	5.81E-01
125-Sb+D [a]	2.47E+05	5.20E+04	1.95E+05	1.30E+01
126-Sn+D [a]	4.64E+02	1.69E+02	2.95E+02	1.96E-02
129-I	1.01E+02	2.20E+01	7.90E+01	5.26E-03
134-Cs	8.71E+04	3.76E+02	8.67E+04	5.77E+00
137-Cs+D [a,c]	6.37E+07	9.11E+05	6.28E+07	4.18E+03
151-Sm	2.61E+06	7.80E+05	1.83E+06	1.22E+02
152-Eu	1.45E+03	3.07E+02	1.14E+03	7.61E-02
154-Eu	1.83E+05	3.77E+04	1.45E+05	9.67E+00
155-Eu	1.76E+05	3.15E+04	1.45E+05	9.62E+00
226-Ra+D [a,b]	6.31E-02	5.70E-02	6.10E-03	4.06E-07
227-Ac+D [a,b]	8.76E+01	6.06E-02	8.75E+01	5.83E-03
228-Ra+D [a,b]	7.71E+01	3.30E+01	4.41E+01	2.94E-03
229-Th+D [a,b]	1.81E+00	3.40E-01	1.47E+00	9.79E-05
231-Pa [b]	1.56E+02	3.44E-01	1.56E+02	1.04E-02
232-Th	4.40E+00	1.28E+00	3.12E+00	2.08E-04
232-U	1.49E+02	3.46E+01	1.14E+02	7.62E-03
233-U	5.72E+02	1.31E+02	4.41E+02	2.94E-02
234-U	3.42E+02	4.41E+01	2.98E+02	1.98E-02
235-U+D [a]	1.46E+01	1.79E+00	1.28E+01	8.53E-04
236-U	1.24E+01	1.43E+00	1.10E+01	7.30E-04
237-Np+D [a,c]	1.85E+02	8.10E+01	1.04E+02	6.92E-03
238-Pu [c]	2.70E+03	1.06E+02	2.59E+03	1.73E-01
238-U+D [a]	3.28E+02	4.83E+01	2.80E+02	1.86E-02

Table 3-3. Estimated HLW Glass Contaminant Concentration.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	Tank Inventory	ILAW Inventory	HLW Inventory [1]	HLW Contaminant Concentration [2]
239-Pu [c]	5.55E+04	3.05E+03	5.25E+04	3.49E+00
240-Pu [c]	1.13E+04	5.25E+02	1.08E+04	7.17E-01
241-Am [c]	1.07E+05	1.08E+04	9.62E+04	6.40E+00
241-Pu	1.66E+05	7.17E+03	1.59E+05	1.06E+01
242-Cm	1.72E+02	5.76E+01	1.14E+02	7.62E-03
242-Pu [c]	1.07E+00	4.49E-02	1.03E+00	6.82E-05
243-Am+D [a,c]	1.76E+01	6.89E-01	1.69E+01	1.13E-03
243-Cm [c]	3.47E+01	6.73E+00	2.80E+01	1.86E-03
244-Cm [c]	7.84E+02	1.01E+02	6.83E+02	4.55E-02
Ag^{+} (silver)	1.51E+03	1.08E+02	1.40E+03	9.33E-02
As^{5+} (arsenic)	2.08E+01	1.76E+01	3.20E+00	2.13E-04
Ba^{2+} (barium)	1.70E+03	1.86E+01	1.68E+03	1.12E-01
Be^{2+} (beryllium)	1.09E+02	6.14E-01	1.08E+02	7.22E-03
Cd^{2+} (cadmium)	4.18E+02	6.30E+01	3.55E+02	2.36E-02
Cl^{-} (chlorine)	9.37E+05	9.31E+05	6.00E+03	3.99E-01
CN^{-} (cyanide)	1.09E+05	0.00E+00	1.09E+05	7.26E+00
Cr (TOTAL)(chromium)	6.72E+05	2.74E+05	3.98E+05	2.65E+01
Cu^{2+} (copper)	3.15E+02	7.33E-01	3.14E+02	2.09E-02
F- (fluoride)	1.20E+06	9.94E+05	2.06E+05	1.37E+01
Fe^{3+} (iron)	1.40E+06	4.48E+04	1.36E+06	9.02E+01
Hg^{2+} (mercury)	2.10E+03	1.92E+02	1.91E+03	1.27E-01
Mn^{4+} (manganese)	1.96E+05	1.38E+04	1.82E+05	1.21E+01
NH_3 (ammonia)	5.01E+05	0.00E+00	5.01E+05	3.34E+01
Ni^{2+} (nickel)	1.80E+05	3.05E+04	1.50E+05	9.95E+00
NO_2^{-} (nitrite)	1.26E+07	0.00E+00	1.26E+07	8.39E+02
NO_3^{-} (nitrate)	5.25E+07	0.00E+00	5.25E+07	3.50E+03

Table 3-3. Estimated HLW Glass Contaminant Concentration.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	Tank Inventory	ILAW Inventory	HLW Inventory [1]	HLW Contaminant Concentration [2]
Pb^{2+} (lead)	8.40E+04	7.83E+03	7.62E+04	5.07E+00
Se^{6+} (selenium)	6.11E-01	5.33E-01	7.80E-02	5.19E-06
SO_4^{2-} (sulfate)	3.91E+06	3.39E+06	5.20E+05	3.46E+01
Tl^{3+} (thallium)	2.54E+04	NA	NA	NA
Zn^{2+} (zinc)	2.89E+03	1.98E+03	9.10E+02	6.06E-02
1,1,1-trichlorethane[d]	NA	0.00E+00	NA	0.00E+00
1,1,2-trichloroethane[d]	NA	0.00E+00	NA	0.00E+00
benzene[d]	NA	0.00E+00	NA	0.00E+00
carbon tetrachloride[d]	NA	0.00E+00	NA	0.00E+00
chloroform[d]	NA	0.00E+00	NA	0.00E+00
ethyl benzene[d]	NA	0.00E+00	NA	0.00E+00
methylene chloride[d]	NA	0.00E+00	NA	0.00E+00
n-butyl alcohol[d]	NA	0.00E+00	NA	0.00E+00
toluene[d]	NA	0.00E+00	NA	0.00E+00
trichloroethylene (1,1,2-trichloroehylene)[d]	NA	0.00E+00	NA	0.00E+00
xylenes-mixed isomers (sum of m-, o-, and p-xylene)[d]	NA	0.00E+00	NA	0.00E+00
1,4-dichlorobenzene[d]	NA	0.00E+00	NA	0.00E+00

[1] HLW inventory equals Tank inventory minus ILAW inventory

[2] HLW concentration equals HLW inventory divided by 15,021 m^3 (from Table 2-2)

[a] The D indicates that the short-lived daughters of these isotopes are in equilibrium with the isotope

[b] These values have been adjusted based on the Kupfer et al. (1999) estimate for tank inventory. Inventories for radionuclides are as of 10/1/98.

[c] Upper bound ILAW inventory estimate based on contract limit

[d] Tank inventories of specific organic compounds are not available; organic compounds are not expected to survive the vitrification process. "NA" indicates components for which inventory information is not available.

The ILAW and HLW glass concentrations are given in Tables 3-1 and 3-3. To estimate the total contaminant inventory in the melters we have assumed 21 HLW melters and 25 LAW melters are disposed in the IDF trench. We have also assumed that each disposed melter

contains the maximum amount of glass (see section 2.2.4). Specifically, each HLW melter contains 5.753 m^3 (5,753 L) of waste form, and each LAW melter contains 14.524 m^3 (14,524 L) of waste form. The total melter inventory for each melter type is provided in Table 3-4.

Table 3-4. Estimated HLW and LAW Inventory in Disposed Melters.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	LAW Average Package Concentration[1]	LAW Melter Inventory [2]	HLW Package Concentration [3]	HLW Melter Inventory [4]
3-H	0.00E+00	0.00E+00	1.64E+00	1.98E+02
14-C	0.00E+00	0.00E+00	2.92E-01	3.52E+01
59-Ni	1.06E-03	3.85E-01	4.60E-02	5.56E+00
60-Co	2.64E-02	9.59E+00	1.05E+00	1.26E+02
63-Ni	1.02E-01	3.70E+01	4.55E+00	5.49E+02
79-Se	3.03E-04	1.10E-01	6.26E-04	7.56E-02
90-Sr+D [a,c]	2.85E+01	1.03E+04	3.69E+03	4.46E+05
93-Zr	7.94E-03	2.88E+00	1.91E-01	2.31E+01
93m-Nb	5.29E-03	1.92E+00	1.13E-01	1.36E+01
99-Tc	1.83E-01	6.64E+01	0.00E+00	0.00E+00
106-Ru+D [a]	5.65E-03	2.05E+00	8.40E+00	1.01E+03
113m-Cd	5.04E-02	1.83E+01	5.81E-01	7.02E+01
125-Sb+D [a]	3.29E-01	1.19E+02	1.30E+01	1.57E+03
126-Sn+D [a]	1.07E-03	3.89E-01	1.96E-02	2.37E+00
129-I	1.39E-04	5.05E-02	5.26E-03	6.35E-01
134-Cs	3.73E-01	1.35E+02	5.77E+00	6.98E+02
137-Cs+D [a,c]	5.76E+00	2.09E+03	4.18E+03	5.05E+05
151-Sm	4.93E+00	1.79E+03	1.22E+02	1.47E+04
152-Eu	1.94E-03	7.04E-01	7.61E-02	9.19E+00
154-Eu	2.38E-01	8.64E+01	9.67E+00	1.17E+03
155-Eu	1.99E-01	7.23E+01	9.62E+00	1.16E+03
226-Ra+D [a,b]	3.61E-07	1.31E-04	4.06E-07	4.91E-05
227-Ac+D [a,b]	3.83E-07	1.39E-04	5.83E-03	7.04E-01

Table 3-4. Estimated HLW and LAW Inventory in Disposed Melters.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	LAW Average Package Concentration[1]	LAW Melter Inventory [2]	HLW Package Concentration [3]	HLW Melter Inventory [4]
228-Ra+D [a,b]	2.09E-04	7.59E-02	2.94E-03	3.55E-01
229-Th+D [a,b]	2.15E-06	7.81E-04	9.79E-05	1.18E-02
231-Pa[b]	2.17E-06	7.88E-04	1.04E-02	1.25E+00
232-Th	8.09E-06	2.94E-03	2.08E-04	2.51E-02
232-U	2.19E-04	7.95E-02	7.62E-03	9.20E-01
233-U	8.26E-04	3.00E-01	2.94E-02	3.55E+00
234-U	2.79E-04	1.01E-01	1.98E-02	2.40E+00
235-U+D [a]	1.13E-05	4.10E-03	8.53E-04	1.03E-01
236-U	9.03E-06	3.28E-03	7.30E-04	8.82E-02
237-Np+D [a,c]	5.13E-04	1.86E-01	6.92E-03	8.36E-01
238-Pu [c]	6.72E-04	2.44E-01	1.73E-01	2.09E+01
238-U+D [a]	3.06E-04	1.11E-01	1.86E-02	2.25E+00
239-Pu [c]	1.93E-02	7.01E+00	3.49E+00	4.22E+02
240-Pu [c]	3.32E-03	1.21E+00	7.17E-01	8.67E+01
241-Am [c]	6.85E-02	2.49E+01	6.40E+00	7.74E+02
241-Pu	4.53E-02	1.64E+01	1.06E+01	1.28E+03
242-Cm	3.64E-04	1.32E-01	7.62E-03	9.20E-01
242-Pu [c]	2.84E-07	1.03E-04	6.82E-05	8.24E-03
243-Am+D [a,c]	4.36E-06	1.58E-03	1.13E-03	1.36E-01
243-Cm [c]	4.26E-05	1.55E-02	1.86E-03	2.25E-01
244-Cm [c]	6.36E-04	2.31E-01	4.55E-02	5.49E+00
Ag^+ (silver)	6.83E-04	2.48E-01	9.33E-02	1.13E+01
As^{5+} (arsenic)	1.12E-04	4.07E-02	2.13E-04	2.57E-02
Ba^{2+} (barium)	1.17E-04	4.25E-02	1.12E-01	1.35E+01
Be^{2+} (beryllium)	3.89E-06	1.41E-03	7.22E-03	8.72E-01
Cd^{2+} (cadmium)	3.98E-04	1.45E-01	2.36E-02	2.86E+00

Table 3-4. Estimated HLW and LAW Inventory in Disposed Melters.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	LAW Average Package Concentration[1]	LAW Melter Inventory [2]	HLW Package Concentration [3]	HLW Melter Inventory [4]
Cl^- (chlorine)	5.89E+00	2.14E+03	3.99E-01	4.83E+01
CN^- (cyanide)	0.00E+00	0.00E+00	7.26E+00	8.77E+02
Cr (TOTAL)(chromium)	1.73E+00	6.28E+02	2.65E+01	3.20E+03
Cu^{2+} (copper)	4.63E-06	1.68E-03	2.09E-02	2.53E+00
F- (fluoride)	6.28E+00	2.28E+03	1.37E+01	1.66E+03
Fe^{3+} (iron)	2.83E-01	1.03E+02	9.02E+01	1.09E+04
Hg^{2+} (mercury)	1.22E-03	4.43E-01	1.27E-01	1.53E+01
Mn^{4+} (manganese)	8.71E-02	3.16E+01	1.21E+01	1.47E+03
NH_3 (ammonia)	2.53E+00	9.19E+02	3.34E+01	4.03E+03
Ni^{2+} (nickel)	1.93E-01	7.01E+01	9.95E+00	1.20E+03
NO_2^- (nitrite)	0.00E+00	0.00E+00	8.39E+02	1.01E+05
NO_3^- (nitrate)	0.00E+00	0.00E+00	3.50E+03	4.22E+05
Pb^{2+} (lead)	4.95E-02	1.80E+01	5.07E+00	6.13E+02
Se^{6+} (selenium)	3.37E-06	1.22E-03	5.19E-06	6.27E-04
SO_4^{2-} (sulfate)	2.15E+01	7.81E+03	3.46E+01	4.18E+03
Tl^{3+} (thallium)	0.00E+00	0.00E+00	NA	NA
Zn^{2+} (zinc)	1.25E-02	4.54E+00	6.06E-02	7.32E+00
1,1,1-trichlorethane[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00
1,1,2-trichloroethane[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00
benzene[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00
carbon tetrachloride[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00
chloroform[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00
ethyl benzene[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00
methylene chloride[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00
n-butyl alcohol[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00
toluene[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00

Table 3-4. Estimated HLW and LAW Inventory in Disposed Melters.

(Inventory in Curies, decayed to October, 1994, for radionuclide and kg for chemical; and concentrations in Ci/m^3 for radionuclide and kg/m^3 for chemical)

Material	LAW Average Package Concentration[1]	LAW Melter Inventory [2]	HLW Package Concentration [3]	HLW Melter Inventory [4]
trichloroethylene (1,1,2-trichloroehylene)[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00
xylenes-mixed isomers (sum of m-, o-, and p-xylene)[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00
1,4-dichlorobenzene[d]	0.00E+00	0.00E+00	0.00E+00	0.00E+00

[1] LAW concentration = ILAW concentration, see Table 3-1

[2] LAW melter inventory based on 25 disposed melters containing 5.753 m^3 of waste

[3] HLW concentration = (tank inventory-ILAW inventory)/15,021 m^3

[4] HLW melter inventory based on 21 disposed melters containing 14.524 m^3 of waste

[a] The D indicates that the short-lived daughters of these isotopes are in equilibrium with the isotope

[b] These values have been adjusted based on the Kupfer et al. (1999) estimate for tank inventory. Inventories for radionuclides are as of 10/1/98.

[c] Upper bound ILAW inventory estimate based on contract limit

[d] Tank inventories of specific organic compounds are not available; organic compounds are not expected to survive the vitrification process. "NA" indicates components for which inventory information is not available.

3.2 PATHWAYS AND SCENARIOS

Relevant pathways and scenarios for these analyses were taken from the 2001 ILAW PA (Mann et al. 2001) and are based on pathways and scenarios used in earlier Hanford Site long-term environmental analysis documents. Five Hanford Site performance assessments for the disposal of low-level waste have already been done (Kincaid et al. 1995, Mann et al. 1998, Wood et al. 1995a, Wood et al. 1995b, and Wood et al. 1996). The most important environmental impact statements (EIS) have been the Hanford Defense Waste EIS (DOE 1987), the Tank Waste Remediation System EIS (DOE 1996), the revised draft for the Hanford Site SWPEIS (DOE/EIS-0286D 2003) and the Final Hanford Comprehensive Land-Use Plan Environmental Impact Statement (DOE 1999c) and its associated record of decision (DOE 1999d). These documents have been fairly consistent in their choice of pathways and scenarios. (Intruder homesteader garden size has decreased from 2500 m^2 to 200 m^2 [see Rittmann 1999].)

The two major exposure pathways considered in this risk assessment are through the use of contaminated groundwater and inadvertent intrusion. These pathways have been shown in previous performance assessments and environmental impact statements to be the dominant pathways. The two major exposure scenarios associated with the use of contaminated groundwater are drinking contaminated water and exposures via all-pathways while living on a small farm (the all-pathways dose). Given the planned future uses of the site the industrial and

residential exposure scenarios are also estimated. The details of these scenarios and the justification for all the parameters used in them are found in Rittmann (1999).

The inadvertent intruder pathway is considered in this risk assessment. The dominant pathway considered is a driller drilling the water well through the disposal site and a future homesteader living on land above the disposal site, using water from the well and raising a small garden for subsistence. The details of these scenarios and the justification for all the parameters used in them are also found in Rittmann (1999).

3.3 VALUES AND ASSUMPTIONS

This section describes and justifies the conceptual models and data for those models that were used in the analyses. It covers the selection criteria and key assumptions for the conceptual models; describes the models and their associated data, the waste form release rates, disposal facility, and moisture infiltration rate. It also covers the dosimetry parameters. The models actually used in the computer simulations were derived from these conceptual models and are described in Section 3.4.

3.3.1 Selection Criteria

The conceptual models and data developed for the 2001 ILAW PA form the basis for the selection of values and assumptions used in this risk assessment for the disposal site. Much experimental and analytical effort has been spent collecting information and producing the understanding needed for the 2001 ILAW PA (Mann et al. 2001). This effort has been documented in a series of data packages. These data packages form the basis for most of the values used in this risk assessment associated with the site, recharge, geochemistry and ILAW-specific parameters (i.e., parameters impacting release rate). Each data package has undergone a hierarchy of reviews. These data packages have been consolidated in Mann and Puigh (2000).

The conceptual models and data associated with the performance of the LLW and MLLW waste forms were largely taken from the SWBG PAs (Wood et al. 1995a and 1996). These specific models for LLW/MLLW contaminant release have been assumed in this risk assessment (see Section 3.4.3.2).

No data exist for the release of contaminants within the HLW and LAW melters.

3.3.2 Key Assumptions

Even though much of the Site-, facility-, and ILAW waste form-specific data needed for this risk assessment have been obtained, some additional assumptions must be made. The key assumptions are as follows:

- The location and layout of the disposal facility, which dictates geology, stratigraphy, infiltration rate, and associated parameters, will not change
- The ILAW waste form composition, which influences the release rate of contaminants from the waste form, will be similar to that currently being proposed
- Our knowledge of tank inventory and the separation and treatment processes used to produce the ILAW packages is adequate

- Our knowledge and understanding of LLW and MLLW inventory and waste release mechanisms from these waste forms is adequate
- Our knowledge and understanding of LAW and HLW melter inventories and waste release mechanisms from these waste forms is adequate
- The disposal facility design will not change significantly
- The well intercept factors (WIFs) calculated for ILAW can be scaled to apply to the IDF trench. (The WIF is defined as the ratio of the contaminant concentration at a well location relative to the contaminant concentration in the aquifer directly beneath the disposal facility.)

The proposed location for the IDF has been identified in the preliminary design report (Comstock and Aromi 2003). However, the layout of the facility on the reserved land may change as design activities proceed.

Preliminary design for the IDF has been initiated (See Section 2.3). Important features have been identified and preliminary investigations have been done (Comstock and Aromi 2003). Thus, certain design features can be included with some confidence.

3.3.3 Disposal Site

Site-specific data for the IDF risk assessment were taken from the significant data collection and analysis activities for the ILAW performance assessment. The following data packages were the sources of these values and assumptions:

- *Near Field Hydrology Data Package for the Immobilized Low-Activity Waste 2001 Performance Assessment* (Meyer and Serne 1999)
- *Geologic Data Packages for 2001 Immobilized Low-Activity Waste Performance Assessment* (Reidel and Horton 1999)
- *Far-Field Hydrology Data Package For The Immobilized Low-Activity Waste Performance Assessment* (Khaleel 1999)
- *Geochemical Data Package For The Immobilized Low-Activity Waste Performance Assessment* (Kaplan and Serne 1999)
- *Groundwater Flow and Transport Calculations Supporting the Immobilized Low-Activity Waste Disposal Facility Performance Assessment* (Bergeron and Wurstner 2000).

Additional data collection has occurred (Mann 2002b) that collaborates and refines the data reported above.

3.3.3.1 Near Field Hydrology. The processes and data important for moisture flow in the zone between the surface and the bottom of the engineered disposal facility are described in *Near-Field Hydrology Data Package for the Immobilized Low-Activity Waste 2001 Performance Assessment* (Meyer and Serne 1999), which is Appendix L in Mann and Puigh (2000). Physical and hydraulic properties (particle size distribution, particle density, bulk density, porosity, water retention, and hydraulic conductivity as a function of moisture content) and associated transport parameters (dispersivity and effective diffusion coefficient) are given for the surface cover materials, the vault structure, diversion layers, the water conditioning layer, and the backfill

materials. Table 3-5 presents best-estimate parameter values for near-field materials. Best estimate values for transport parameters (which are relatively unimportant in this analysis) can be found in Meyer and Serne (1999, Chapter 5).

Table 3-5. Best-Estimate Hydraulic Parameter Values For Near-Field Materials.

Material	ρ_p (g/cm^3)	ρ_b (g/cm^3)	θ_s	θ_r	α (cm^{-1})	n	K_s (cm/s)
Surface Barrier							
Silt loam-gravel admixture	2.72	1.48	0.456	0.0045	0.0163	1.37	$8.4x10^{-5}$
Compacted silt loam	2.72	1.76	0.353	0.0035	0.0121	1.37	$1.8x10^{-6}$
Sand filter	2.755	1.88	0.318	0.030	0.538	1.68	$8.58x10^{-5}$
Gravel filter	2.725	1.935	0.290	0.026	8.1	1.78	$1.39x10^{-2}$
Gravel drainage	2.725	1.935	0.290	0.006	17.8	4.84	2.0
Asphaltic concrete	2.63	2.52	0.04	0.000	$1.0x10^{-7}$	2.0	$1x10^{-11}$
Capillary Break							
Diversion layer sand	2.8	1.65	0.371	0.045	0.0683	2.08	$3.00x10^{-2}$
Diversion layer gravel	2.8	1.38	0.518	0.014	3.54	2.66	1.85
Trench/Vault							
Filler material	2.63	1.59	0.397	0.005	0.106	4.26	$3.79x10^{-2}$
Glass waste	2.68	2.63	0.02	0.00	0.2	3	0.01
Backfill	2.76	1.89	0.316	0.049	0.035	1.72	$1.91x10^{-3}$

ρ_p = particle density; ρ_b = dry bulk density
θ_s = saturated water content; θ_r = residual water content
α,n = van Genuchten fitting parameters; K_s = saturated hydraulic conductivity

3.3.3.2 Geology. Reidel and Horton (1999) summarizes the stratigraphy data used in this risk assessment. The key elements are the major units and their location in the vadose zone beneath the proposed site. The material types beneath the disposal site have been grouped into two types: Hanford sands and the Hanford gravelly formation. Figure 2-2 shows the position of these materials beneath the proposed site along a west-east cross section. If we assign a coordinate system with Z=0 at the ground surface centerline of the disposal site, then the upper gravel sequence extends ~-7 meters below the surface. From Z=-7 to ~-79 m, the material is assigned the hydraulic properties corresponding to Hanford sands (see Section 3.3.3.3). From Z=-79 m to ~-103 m, the material is assigned the hydraulic properties corresponding to the Hanford gravelly formation. From Figure 2-2 we see that the interface between these two layers slopes upward from directly beneath the center of the disposal site toward the west. At the western edge of the disposal site the depth to this interface is approximately Z=-75 m.

3.3.3.3 Far Field Hydrology. Khaleel (1999) summarizes the upscaled hydraulic parameter estimates based on data from the ILAW borehole and data on gravelly samples from the 100 Area boreholes. Table 3-6 provides the best estimate (or mean) values affecting moisture flow. Khaleel (1999) describes the processes for upscaling small-scale laboratory measurements to field-scale applications, and provides recommendations for determining which parameters to use at that scale. Best estimate values for transport parameters associated with the effective transport parameters (bulk density, diffusivity, and dispersivity) also are described in Khaleel (1999).

Table 3-6 Best-Estimate Hydraulic Parameter Values For Far-Field Layers.

Formation	θ_s	θ_r	α (1/cm)	n	P	K_s (cm/s)
Sandy	0.375	0.041	0.057	1.768	0.5	2.88×10^{-3}
Gravelly	0.138	0.010	0.021	1.374	0.5	5.60×10^{-4}

θ_s = saturated water content θ_r = residual water content
α, n = van Genuchten fitting parameters P = pore size distribution factor
K_s = saturated hydraulic conductivity

Overall, compared to the sandy sequence, the gravelly sequence is characterized by a much smaller saturated water content, higher bulk density, higher log-conductivity variance, smaller log-unsaturated conductivity variance, a much smaller macroscopic anisotropy and smaller dispersivities (Khaleel 1999). An anisotropy ratio (ratio of horizontal to vertical hydraulic conductivity) in excess of one results in an enhanced lateral migration. To minimize lateral migration (i.e., a conservative assumption), an isotropic model was used for both strata.

Longitudinal dispersivities of 200 cm and 30 cm were used for the sandy and gravelly sequences, respectively (Khaleel 1999). Lateral dispersivities were estimated to be $1/10^{th}$ of the longitudinal estimates. The effective, large-scale diffusion coefficients for both sandy and gravel-dominated sequences are assumed to be a function of volumetric moisture content, θ. The computer code, VAM3DF (Huyakon and Panday 1999) is used to calculate the fluid and contaminant transport in the far field. VAM3DF uses the Millington and Quirk (1961) empirical relation:

$$D_e(\theta) = D_0 \frac{\theta^{10/3}}{\theta_s^2} \tag{3.1}$$

where

$D_e(\theta)$ is the effective diffusion coefficient of an ionic species

D_0 is the effective diffusion coefficient for the same species in free water

θ is the volumetric moisture content, and

θ_s is the saturated volumetric water content.

The molecular diffusion coefficient for all species in pore water is assumed to be 2.5 x 10^{-5} cm^2/s (Kincaid et al. 1995).

3.3.3.4 Geochemistry. Chemical interactions with the facility, near-field materials, and the soil in the vadose zone can greatly slow the transport of contaminants. Geochemical effects are based on the discussion and values presented in *Geochemical Data Package for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment* (Kaplan and Serne 1999), also provided in Appendix N of Mann and Puigh (2000).

The geochemistry is described using the distribution coefficient (K_d value) of a specified solid. The distribution coefficient is a thermodynamic construct. It is the ratio of the concentration of a species reversibly adsorbed or exchanged to a geomedia's surface site divided by the concentration of the species in solution. Parameters are given in Kaplan and Serne (1999) for the following three zones:

- Near-Field. Inside the disposal facility (K_d and solubility values)
- Chemically Impacted Far-Field in Sand Sequence. (K_d values only)
- Chemically Impacted Far-Field in Gravelly Sequence. (K_d values only)

The amount of slowing is described by a multiplicative factor known as the geochemical retardation factor, which involves the distribution coefficient. Geochemical retardation in unsaturated conditions is predicted to be

$$R_f = 1 + \rho K_d / \theta \qquad (3.2)$$

where

R_f is the geochemical retardation factor (dimensionless)
ρ is the bulk density of the material (g/cm^3)
K_d is the chemical distribution coefficient (liter/g)
θ is the volumetric moisture content (dimensionless).

A derivation of the general contaminant transport equation is given in the 1998 ILAW PA report (Mann et al. 1998, Appendix D, Section D.2.3). The chemical distribution coefficient (K_d) is measured in the laboratory by comparing the amount of material trapped in or on the soil matrix to the amount of material in the water phase.

Table 3-7 provides estimates for K_d from recent measurements and for the K_ds used in the analyses in this report. Unless otherwise stated, the K_ds are provided for the chemically impacted far-field sandy sequence beneath the disposal facility (Table 3-7). The "Probable K_d" is the best estimate for the K_d. Finally, the "K_d value used" refers to the value of K_d used in the analyses provided in this report.

For convenience in modeling, a subset of K_d values was used in these analyses. The computer code VAM3DF (Huyakon and Panday 1999) treats the chemical distribution coefficients as point-estimate values, not as probability functions. Therefore, the actual K_d values used were reduced to one of eight value sets for the far field (see Tables 3-7). This K_d value was conservatively chosen to be one of the following six values: 0, 0.6, 4.0, 10, 80, and 150 mL/g.

These values are less than or equal to the probable K_d value provided in Table 3-7. The elements selected were shown to be the most important in the 1998 ILAW PA. The values in

parentheses provided in Table 3-7 are for the unperturbed (near neutral pH, ionic strengths between 0 and 0.01, and only trace contaminant concentrations) far-field sand sequence.

Because radionuclides spend significantly less time in the unconfined aquifer than in the vadose zone, no credit was taken in this analysis for increased travel time in the unconfined aquifer because of geochemical retardation.

Values are based on site-specific samples for the most part, but in a few cases depend on literature values or chemical similarity. Table 3-7 provides the best estimate K_d values for the chemically impacted far-field sand sequence. The gravel-corrected best estimate K_d values for the chemically impacted far-field gravel sequence are a factor of 10 smaller than the values given in Table 3-7. The aqueous phase is assumed to be untainted Hanford formation groundwater except for trace levels of radionuclide and the solid phase is assumed to be natural Hanford formation sand-dominated sequence sediment. The literature values on which these values were based had an aqueous phase near neutral pH, ionic strength between 0 and 0.01, and trace radionuclide concentrations. For contaminants where no K_d value is provided in Table 3-7, a K_d value of 0 was assumed.

Table 3-7. Best-Estimate K_d Values For The Far-Field Sand Sequence.

Radionuclide	Probable K_d [a,b] (mL/g)	Value Used [a,c] (mL/g)
Ac	350.	150.
Am	350.	150.
C [(d)]	20. (5.)	4.
Ce	350.	150.
Cl	0.	0.
Cm	350.	150.
Co	300.	150.
Cs	80.	80.
Eu	350.	150.
^3H	0.	0.
I	0.	0.
Nb	80.	80.
Ni	80.	80.
Np	0.8	0.6
Pa	0.8	0.6
Pb	100.	80.
Pu	200.	150.
Ra	10.	10.
Ru	1.	0.6
Se	4.	4.
Sn	80.	80.
Sr	10.	10.
Tc	0.	0.
Th	300.	150.

Table 3-7. Best-Estimate K_d Values For The Far-Field Sand Sequence.

Radionuclide	Probable K_d [a,b] (mL/g)	Value Used [a,c] (mL/g)
U [(d)]	10. (0.6)	0.6
Zr	300.	150.

[a] The values in the table are for the chemically impacted far-field sand sequence. The aqueous phase is moderately altered from the cement and glass leachate emanating from the near field; pH is between 8 (background) and 11, and the ionic strength is between 0.01 (background) and 0.1. The solid phase is in the sand-dominated sequence and is slightly altered because of contact with the caustic aqueous phase.

[b] Probable K_d is the best estimate for K_d

[c] Value Used is the K_d value used in the analyses provided in this report for the sandy sequence. For the gravelly sequence the K_d value used is 0.1 times the value indicated in the table

[d] The values in parentheses in the table are for the unperturbed far-field sand sequence. The aqueous phase is assumed to be untainted Hanford formation groundwater, except for trace levels of radionuclide and the solid phase is assumed to be natural Hanford formation sand-dominated sequence sediment. The literature values on which the values were based had an aqueous phase near neutral pH, ionic strength between 0 and 0.01, and trace radionuclide concentrations.

3.3.3.5 Unconfined Aquifer. Groundwater flow and contaminant transport were calculated with the Hanford Sitewide Groundwater model (as defined and used in Bergeron and Wurstner 2000). This three-dimensional model, currently being used by the Hanford Groundwater Project and recommended as the proposed Sitewide groundwater model in the Hanford Site groundwater model consolidation process, is based on the Coupled Fluid, Energy, and Solute Transport (CFEST-96) Code (Gupta et al. 1987). The specific implementation of this model is more fully described in Wurstner et al. 1995 and Cole et al. 1997. This specific model was most recently used in the Hanford Site Composite Analysis (Cole et al. 1997; Kincaid et al. 1998), which is a companion analysis to the 1998 performance assessment analyses of the ILAW disposal (Mann et al. 1998) and the solid waste burial grounds in the 200 East and 200 West Areas (Wood et al. 1995a and 1996).

Bergeron and Wurstner (2000) predicted the water table for the post-Hanford steady state conditions to be at an elevation of between 117 and 118 meters above mean sea level. From the geologic information from the nearby boreholes (see Figure 2-2) the estimated groundwater level is approximately 103 meters below the ground surface at the IDF site.

A local scale model was developed for the 2001 ILAW PA and well intercept factors (WIFs) were determined for representative well locations along projected flow lines for the unconfined aquifer. The WIF is defined as the ratio of the contaminant concentration at a downgradient well location relative to the contaminant concentration in the aquifer directly beneath the disposal facility. These WIFs are documented in the *Groundwater Transport Calculations Supporting the Immobilized Low-Activity Waste Disposal Facility Performance Assessment* (Bergeron and Wurstner 2000).

3.3.4 Waste Package and Waste Form Release

This section describes the conceptual models and data used to represent the different waste packages within the scope of this risk assessment. These waste packages include the ILAW waste package; the waste packages associated with LLW/MLLW and the HLW and LAW melter waste packages.

3.3.4.1 ILAW. The DOE intends to process approximately 10% of the waste from the Hanford tanks in an initial phase (Phase 1). The definition of the product form and specification for the remaining 90% of the Hanford tank waste are not defined at this time. For the purposes of this assessment activity, all the ILAW waste products are assumed equivalent to the DOE specifications for the Phase 1 contract and current plans.

Modification 12 of the BNFL contract (see DOE/BNFL 1998), which was issued on January 24, 2000, and the current contract with Bechtel Washington (DOE/ORP 2000) require ILAW canisters in the form of right circular cylinders (1.22 m diameter by 2.29 m tall).

For this risk assessment we have assumed these waste packages to be filled to a height of 2.0 m and backfilled with inert material.

For the waste form calculations discussed in Section 3.4.3, the glass waste material was assumed to be fractured. Also, the surface area was assumed to be 10 times greater than that of an unfractured right circular cylinder of diameter 1.22 m and 2 m in height.

The 2001 ILAW PA (Mann et al. 2001) showed that the release rate from the waste form was one of the key parameters in the performance assessment. This rate is a major determinant of the impact of disposal as well as setting the temporal structure of that impact. The data for determining the waste form release rate are given in *Waste Form Release Data Package for the 2001 Immobilized Low-Activity Waste Performance Assessment* (McGrail et al. 2001) and Appendix K of Mann and Puigh (2000). The details of the conceptual model and calculational approach for estimating the contaminant release rate from the ILAW are documented in Bacon and McGrail (2002). The 2001 ILAW PA showed that the container does not affect the release rate from the ILAW waste form. Thus, in these calculations the container is ignored.

3.3.4.2 LLW/MLLW. The majority of LLW requires no waste form or waste package performance. MLLW packages are often macro encapsulated in grout or plastic to satisfy land disposal restriction requirements that are loosely associated with long-term control of contaminant release. The use of waste packages and waste forms to control contaminant releases to the environment is generally limited to wastes containing relatively large inventories of environmentally mobile constituents (e.g., Tc-99, I-129 and uranium). Waste package/waste form performance is generally provide by Portland cement-based concrete containers or grouted waste. These materials and disposal configurations act either as a chemical buffer (in the case of uranium) to reduce solubility or as a diffusion barrier to limit releases to the environment (e.g., high Tc-99 or I-129 inventory wastes).

3.3.4.3 WTP Melters. The final HLW and LAW melter and overpack designs have not been completed. Preliminary design and planning information have been used to estimate waste package materials, dimensions, and quantities. For this risk assessment we assume carbon steel overpacks are used for both the HLW and LAW melters. The dimensions of these overpacks are given in Table 2-3. The HLW melter overpack is assumed to have a rectangular geometry with 8-inch thick steel walls. The LAW melter overpack is assumed to have 1-inch thick steel walls.

The number of melters requiring disposal for the balance of the WTP mission is estimated to be 46 melters, including 18 HLW melters, 22 LAW melters, and 6 contingency melters. Each melter is assumed to contain the maximum amount of glass possible. From Section 2, the final maximum glass volume remaining in a HLW melter is 5,753 L (351,378 cubic inches) or 1.50×10^4 kg and the final maximum glass volume remaining in a LAW melter is 14,524 L (887,112 cubic inches) or 3.8×10^4 kg.

The release mechanism associated with the HLW and LAW waste mixtures in the melters is not known. We anticipate that the residual material will be predominantly glass-like. We have also assumed that both the LAW and HLW melters are grouted within their respective overpacks and the release rate is bounded by the diffusion of the contaminant through the grout. The diffusion coefficient used in this risk assessment is discussed in Section 3.4.3.2.2.

3.3.5 Disposal Facility

The IDF trench preliminary design summarized in Section 2.2 is used for the calculations. The dimensions for the IDF trench model are taken from Figure 2-4.

The key components of the disposal system are the surface barrier, the sand-gravel capillary break, the trench and the filler material. The surface barrier design has not yet been established and is assumed to be a modified RCRA-compliant subtitle C cap as described in Puigh 1999 (Section 4.0). Note that the cap is shaped like an inverted "v" and placed with its apex along the length dimension (north-south) and centered over the trench. The slope of the cap is assumed to be 2%. The cap is assumed to extend 10 m beyond the inside edge of the trench (dimension taken from conceptual design report for ILAW Disposal Facility [Burbank et al. 2000]). This cap includes an asphalt layer and has a design life of 500 years. Beneath the surface cap is a sand-gravel capillary break. The sand layer is assumed to be 1 meter thick. A gravel layer is built up at the apex and with a 2% slope to support the surface cap. The cap height was chosen to ensure that the waste packages are greater than 5 meters below the surface (per 10 CFR 61 requirements).

The trench dimensions are as defined in Section 2.2. The leachate collection systems are ignored in the moisture and transport modeling. The leachate collection systems can be ignored because of the relatively short design life for these material (less than 100 years for HDPE) compared to the travel time through the vadose zone (1,000-2,000 years). The 1998 ILAW PA (Mann et al. 1998) and the 2001 ILAW PA (Mann et al. 2001) examined the potential impact of the concrete vault trapping water and then failing ("bathtub effect"). These analyses showed little effect on the estimated impacts at the time of compliance. The material between the packages in the trench is assumed to be backfill material as defined in Meyer and Serne 1999.

The model calculations for the release of contaminants from the waste packages to the groundwater and the estimated impacts associated with the intruder scenarios depend on the waste loading of the various waste packages into the IDF trench. For all waste types, we have

assumed the packing fraction of waste packages into the IDF trench results in a 40% by volume waste package loading into the available trench volume. The top of the trench and the upper HPDE liner determines the available trench volume. The remaining fill material into the trench is assumed to be backfill.

For the ILAW waste packages, the maximum stacking height is 4 layers of ILAW waste packages. This corresponds to a maximum glass height of 8 meters within a given position within the trench.

For the LLW/MLLW waste packages, we have assumed the maximum stacking height of waste within the trench is proportional to the maximum stacking height used in the SWBG PAs (Wood et al. 1995a and 1996). For these PAs the waste stacking height was 5 meters within a 7-meter deep trench. Since the IDF trench is 13.2 meters deep, the maximum stacking height for the LLW/MLLW is estimated to be 9.4 (=13.2*5/7) meters.

For the HLW and LAW melters, a maximum of 2 melter overpacks can be stacked on top of each other. This is based on the estimated heights for the melter overpacks (see Table 2-3), the assumption of a minimum operations layer of 0.9 m between stacked waste packages, and the maximum available depth for the trench (13.2 m). From Section 2.2.3 the maximum glass height within one HLW melter is 1.09 m, and the maximum glass height within one LAW melter is 0.97 m. Therefore the maximum glass height for two melters stacked on top of each other is 2.18 m (assuming 2 HLW melters stacked on top of each other) within the IDF trench.

3.3.6 Waste Package Loading into IDF Trench

The waste package loading into the IDF trench determines the area over which the contaminants are distributed and the size of the IDF trench at closure. This area is determined by the waste package volume fraction loaded into the trench and the volume of waste for each category type that is disposed in the IDF trench. Volume estimates have been provided for the various waste types (ILAW, LLW, MLLW, and melters). Also the waste package volume fraction loaded into the IDF trench is assumed to be 40% by volume. Finally, the current operational plan for filling the IDF is to use one-half the trench for ILAW and MLLW (including melters) and the other one-half for the LLW.

Table 3-8 summarizes the nominal and upper bound waste package volume estimates and the corresponding trench volumes needed to accommodate the different waste forms. The bottom cell length is calculated assuming a sloped region on only one side of each cell. The effective area is the surface area of the cell region containing the waste. From the total nominal and upper bound waste volumes a corresponding equivalent trench length can be calculated. This length assumes a rectangular shape for the IDF trench and corresponds to the north-south length along the bottom of the IDF trench. The total area provided in the table is the total surface area of the trench required to contain the IDF inventory.

Table 3-8. Estimated IDF Trench Size Required to Contain Inventory.

Waste Form	Waste Package Volume (m^3)			Minimum Trench Vol. Needed (m^3)	Bottom Cell Length (m)	Effective Surface Area (m^2)
	# of packages	Package Vol. (m^3)	Vol. (m^3)			
ILAW - nominal	70,064	2.69	188,472	471,180	174.3	39,146
ILAW - upper bound	81,000	2.69	217,890	544,725	201.5	45,257
LLW (Cat. 1) - nominal	--	--	103,000	257,500	95.3	21,394
LLW (Cat. 3) - nominal	--	--	8,000	20,000	7.40	1,662
MLLW - nominal	--	--	53,500	133,750	49.5	11,112
LLW - upper bound	--	--	306,000	765,000	283.0	63,558
MLLW - upper bound	--	--	188,000	470,000	173.9	39,048
HLW Melters	21	121.9	2,560	6,400	2.4	532
LAW Melters	25	299.4	7,485	18,713	6.9	1,555
				Total Volume (m^3)	Equivalent Trench Bottom Length (m)	Total Area (m^2)
Nominal Inventory Trench parameters				907,543	167.9	75,400
Upper Bound Inventory Trench parameters				1,804,837	333.8	149,949

3.3.7 Infiltration or Recharge Rate

The term recharge is used to denote the rate at which moisture flows past the root zone (that is, very near surface) into a region where moisture flow follows simpler models. Recommendations for recharge rates are taken from *Recharge Data Package for the Immobilized Low-Activity Waste 2001 Performance Assessment* (Fayer 1999), and are also provided in Appendix J of Mann and Puigh (2000). Long-term estimates of moisture flux through a fully functional surface cover, the cover side slope, and the immediate surrounding terrain, as well as for degraded cover conditions are needed. These estimates were derived from lysimeter and tracer measurements collected by the ILAW PA activity and by other projects combined with a modeling analysis.

Values for the recharge are given in Table 3-9. Values are given for two separate surface soils, Rupert sands and Burbank loamy sands. The Rupert sands are located at the site of the existing grout vaults and at the southernmost 60% of the new ILAW disposal site. The Burbank loamy sand is located at the northernmost 40% of the new ILAW disposal site. Impacts from degradation of the surface barrier, vegetation change, climate change, and irrigation were considered in establishing the best estimate and bounding values.

Table 3-9. Recharge Rate Estimates (mm/year).[a]

Surface feature	Pre-Hanford	Construction	Cover and Post Cover Design Life
Surface cover	na	na	0.1 (0.01, 4.0)
Cover side slope	na	na	50 (4.2, 86.4)
Rupert sand	0.9 (0.16, 4.0)	0.9 (0.16, 4.0)	0.9 (0.16, 4.0)
Burbank loamy sand	4.2 (2.8, 5.5)	4.2 (2.8, 5.5)	4.2 (2.8, 5.5)
Construction	na	55.4 (50, 86.4)	na

[a] Best estimate case given, with values for reasonable bounding cases given in parentheses; na = not applicable

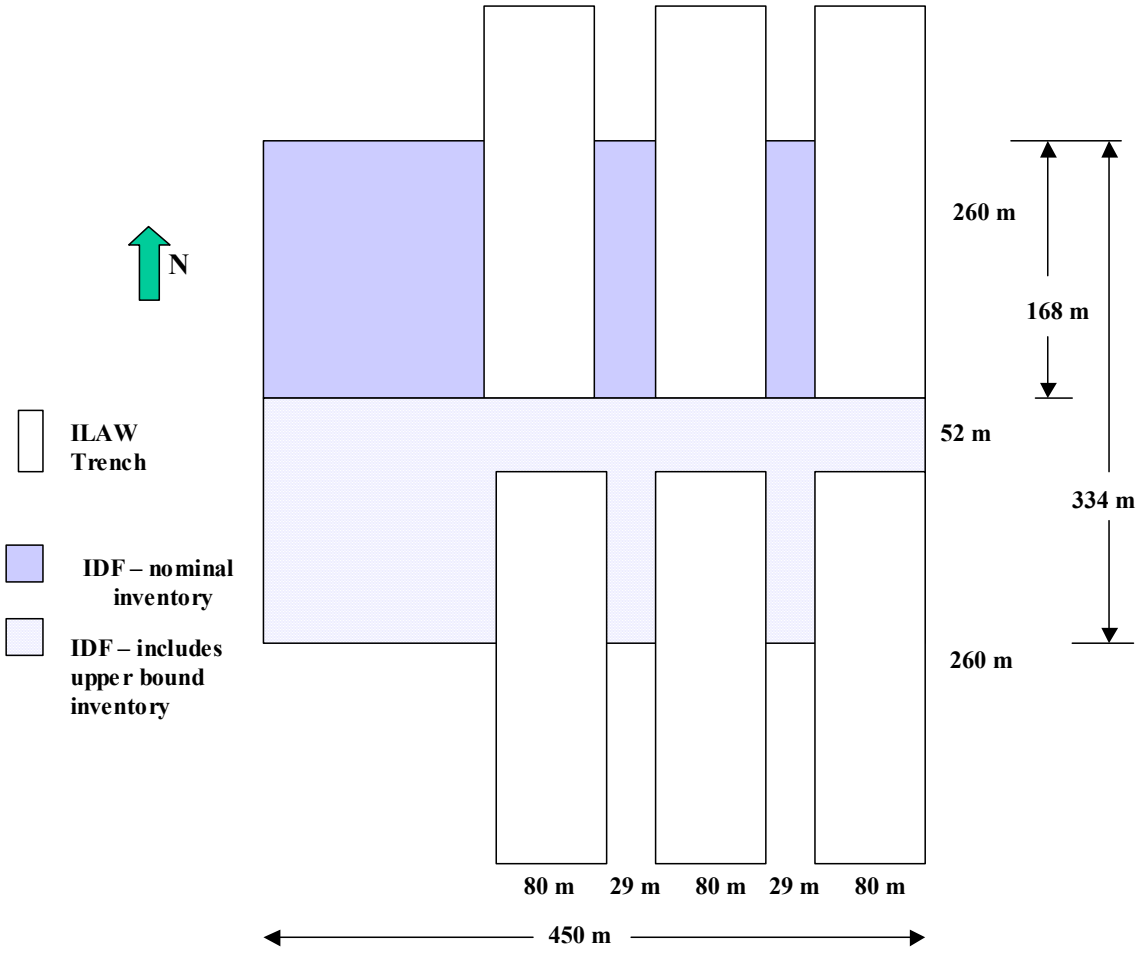
3.3.8 Well Intercept Factors

The well intercept factor (WIF) used in this risk assessment is based on the data and modeling work that formed the basis for the well intercept factors used in the 2001 ILAW PA (Bergeron and Wurstner 2000). In that assessment the Hanford Site-Wide Groundwater Model was applied to the ILAW disposal site. The site-wide model and supporting local scale models were used to evaluate the mixing of contaminated water from the disposal facility into the underlying aquifer and the subsequent lateral migration of contaminants to receptor points and/or points of groundwater discharge along the Columbia River. The estimated WIFs from Bergeron and Wurstner (2000) assume a unit contaminant flux uniformly distributed within the areal distribution of the six-trench configuration representing the remote handled trench concept (Puigh 1999) assumed in the 2001 ILAW PA. The WIFs were calculated for a range of recharge rates, different locations within the disposal site and different orientations of the trenches with respect to the groundwater flow. Bergeron and Wurstner (2000) showed the WIFs were insensitive to the pumping rate at the location 100 m downgradient from the disposal facility.

Figure 3-1 shows a comparison of the footprint for the current disposal facility compared to the footprint of the six-trench configuration used in developing the WIF for the 2001 ILAW PA. The footprint of the IDF for nominal waste loading is smaller than that of the six-trench configuration.

This would tend to decrease the estimated WIF for the IDF when compared to the WIF used in the 2001 ILAW PA. From the sensitivity studies performed (Bergeron and Wurstner 2000) two observations were made: 1) the WIFs were proportional to the disposal trench area for the sensitivity cases investigated and 2) the WIF increased as the trenches were moved from the northern to the southern portion of the disposal site.

Figure 3-1. Comparison of IDF Footprint to the ILAW Remote Handled Trench Footprint Used in the 2001 ILAW PA.



For this risk assessment we have assumed that the WIF for the IDF scales as the ratio of the area between the six-trench configuration (6 x 80m x 260m = 124,800 m^2 and the IDF area at the top of the trench. From Table 3-8 the IDF area is 75,400 m^2 for the nominal inventory loading and 149,949 m^2 for the upper bound inventory loading. Therefore, the IDF WIF is a factor of 0.604 (75,400/124,800) times smaller than the ILAW WIF at the same location within the ILAW site for the nominal inventory loading and 1.20 times larger than the ILAW WIF for the upper bound inventory. We have assumed the ILAW WIF at the south end of the ILAW disposal site to be more representative of the IDF. Therefore, the proposed IDF WIFs are summarized in Table 3-10.

Table 3-10. WIF for Selected Cases.

Cases	WIF @ Infiltration Rate of 4.2 mm/y		
	100 m	1000 m	Columbia River [1]
ILAW (Base Case) [a]	1.1×10^{-3}	7.8×10^{-4}	9.8×10^{-5}
ILAW (Trenches at South End of Site) [a]	2.0×10^{-3}	1.1×10^{-3}	9.8×10^{-5}
IDF – Nominal Inventory	1.2×10^{-3}	6.6×10^{-4}	5.9×10^{-5} [b]
IDF – Upper Bound Inventory	2.4×10^{-3}	1.3×10^{-3}	1.2×10^{-4}

[a] From Bergeron and Wurstner 2000
[b] WIF calculated using Hanford Site Model; assume WIF just before the Columbia River (CR) unchanged if trenches at south end of site; assume CR WIF scales with area for the IDF.

To first order, the amount of groundwater dilution is linearly dependent on the groundwater flow. The *Hanford Site Groundwater Monitoring for Fiscal Year 2002 report* (Hartman et al. 2003) presents in Appendix A, Table A.2 current estimates for groundwater flows in the Hanford 200 East and 200 West Areas. For example, the groundwater flow for Low-Level Waste Management Area 3 is estimated to be 0.0001 to 0.12 meters/day. Other facilities (such as the tank farms) in the 200 West Area have similar groundwater flow rates (0.003 to 0.36 m/d). In contrast, groundwater flow rates typical of the IDF site (which is not given in the report) are much higher. For example, the groundwater flow rate at the A/AX Tank Farms (the tank farms nearest the IDF site) is estimated at 1.6 to 3.0 meters/day. Such higher flow is a result of the presence of the ancient Columbia River channel through the eastern edge of the 200 East Area.

3.3.9 Exposure Parameters

Dosimetry scenarios and parameter values are based on the discussion and values presented in *Exposure Scenarios And Unit Dose Factors for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment* (Rittmann 1999), and also Appendix O of Mann and Puigh (2000). The scenarios for human exposure to the hazardous materials associated with the ILAW glass are defined in appendix B (Mann 1999b). Table 3-11 provides the unit dose factors (mrem per Ci exhumed) for the intrusion scenario where a post-intrusion resident lives near the exhumed waste associated with a well drilled through the disposal site. Table 3-12 provides the total unit dose factors for five exposure scenarios where the exposure includes contamination of the groundwater. These scenarios are for industrial, residential, agricultural, and population exposures as defined in the Hanford Site Risk Assessment Methodology (HSRAM) (DOE/RL 1991). The Native American subsistence resident exposure is discussed in DOE/RL 1997.

In the *Evaluation of the Potential for Agricultural Development at the Hanford Site* (Evans et al. 2000), well screen heights in the local tri-county area were surveyed. The continued use of the 4.6-meter (15 foot) well screen height is justified, given that most screen heights are larger than this value.

Estimates for the ILCR and HI associated with chromium, nitrate, and uranium were developed from the latest IRIS database (EPA 2003) and are provided in Table 3-13 for the all-pathway farmer scenario.

Table 3-11. Annual Unit Dose Factors for Post-Intrusion Resident (mrem per Ci exhumed).[a]

Radionuclide	External	Internal	Radionuclide	External	Internal
H-3	0.0	$1.46x10^{2}$	U-234	$9.04x10^{-1}$	$2.68x10^{3}$
Se-79	$4.24x10^{-2}$	$1.24x10^{2}$	U-235+D	$1.66x10^{3}$	$2.51x10^{3}$
Sr-90+D	$5.15x10^{1}$	$2.00x10^{4}$	U-236	$4.81x10^{-1}$	$2.54x10^{3}$
Tc-99	$1.69x10^{-1}$	$7.93x10^{2}$	U-238+D	$2.61x10^{2}$	$2.45Ex10^{3}$
Sn-126+D	$2.41x10^{4}$	$1.05x10^{2}$	Np-237+D	$2.30x10^{3}$	$2.39x10^{4}$
I-129	$2.58x10^{1}$	$6.70x10^{3}$	Pu-239	$6.48x10^{-1}$	$1.18x10^{4}$
Cs-137+D	$6.80x10^{3}$	$1.23x10^{3}$	Pu-240	$3.34x10^{-1}$	$1.18x10^{4}$
Pa-231	$4.78x10^{2}$	$3.81x10^{4}$	Am-241	$9.98x10^{1}$	$1.23x10^{4}$
U-233	3.21	$2.74x10^{3}$			

[a] Dose factors are based on 50-year total effective dose equivalent resulting from intakes during the first year after intrusion.

Table 3-12. Total Annual Unit Dose Factors for Low-Water Infiltration Cases (mrem per pCi/L in the groundwater).[a]

Nuclide	HSRAM Industrial[b]	HSRAM Residential[b]	All Pathways Farmer[b]	Drinking Water (All Pathway Farmer)[b]	Native American Subsistence Resident[b]
H-3	$1.62x10^{-5}$	$4.92x10^{-5}$	$4.58x10^{-5}$	$3.46x10^{-5}$	$1.03x10^{-4}$
Se-79	$2.18x10^{-3}$	$7.26x10^{-3}$	$1.15x10^{-2}$	$4.70x10^{-3}$	$3.10x10^{-2}$
Sr-90+D	$3.83x10^{-2}$	$1.30x10^{-1}$	1.19E-01	$8.26x10^{-2}$	$3.38x10^{-1}$
Tc-99	$3.65x10^{-4}$	$1.31x10^{-3}$	$3.54x10^{-3}$	$7.88x10^{-4}$	$1.23x10^{-2}$
Sn-126+D	$5.28x10^{-3}$	$4.07\ x10^{-2}$	$5.63x10^{-2}$	$1.14x10^{-2}$	$1.20x10^{-1}$
I-129	$6.90x10^{-2}$	$2.31x10^{-1}$	$3.77x10^{-1}$	$1.49x10^{-1}$	1.21
Cs-137+D	$1.25x10^{-2}$	$4.84x10^{-2}$	$7.53x10^{-2}$	$2.70x10^{-2}$	$2.14x10^{-1}$
Pa-231	2.68	8.87	7.08	5.72	1.84E+01
U-233	$7.51x10^{-2}$	$2.45x10^{-1}$	$2.19x10^{-1}$	$1.56x10^{-1}$	$5.77x10^{-1}$
U-234	$7.35x10^{-2}$	$2.40x10^{-1}$	$2.14x10^{-1}$	$1.53x10^{-1}$	$5.65x10^{-1}$

Table 3-12. Total Annual Unit Dose Factors for Low-Water Infiltration Cases (mrem per pCi/L in the groundwater).[a]

Nuclide	HSRAM Industrial[b]	HSRAM Residential[b]	All Pathways Farmer[b]	Drinking Water (All Pathway Farmer)[b]	Native American Subsistence Resident[b]
U-235+D	$6.93x10^{-2}$	$2.28x10^{-1}$	$2.03x10^{-1}$	$1.44x10^{-1}$	$5.34x10^{-1}$
U-236	$6.99x10^{-2}$	$2.28x10^{-1}$	$2.04x10^{-1}$	$1.45x10^{-1}$	$5.37x10^{-1}$
U-238+D	$6.95x10^{-2}$	$2.27x10^{-1}$	$2.03x10^{-1}$	$1.45x10^{-1}$	$5.34x10^{-1}$
Np-237+D	1.12	3.72	2.97	2.40	7.73
Pu-239	$8.94x10^{-1}$	2.96	2.36	1.91	6.14
Pu-240	$8.94x10^{-1}$	2.96	2.36	1.91	6.14
Am-241	$9.19x10^{-1}$	3.05	2.43	1.97	6.32

[a] Dose factors are based on 50-year total effective dose equivalent resulting from intakes during the first year of irrigation.
[b] Annual dose in mrem for a groundwater concentration of 1 pCi/L.

Table 3-13. Risk Factors Associated with the All Pathways Farmer Scenario.[a]

Chemical	Hazard Index per mg/L	Incremental Lifetime Cancer Risk Risk per mg/L
Cr^{6+} (chromium)	$1.81x10^{1}$	$9.13x10^{-5}$
NO_3 (nitrate)	$6.3x10^{-2}$	NA[b]
Uranium (Total)	$9.0x10^{1}$	NA[b]

[a] Based on exposure to well water only
[b] NA = not applicable

3.4 RISK ASSESSMENT METHODOLOGY

This section describes how the performance of the system was established. That is, this section explains how the data and conceptual models presented in Sections 3.1 through 3.3 are translated into a numerical model suitable for computer simulation. First, the strategy of the computer simulation is introduced. Next the process of translating the disposal facility concepts and the natural system into computer models is described. Finally, the parameters used in the computer simulations are given.

Calculations of relatively simple equations for gaseous diffusion were done by hand. These equations will be treated in Chapter 4, where the results are discussed.

3.4.1 Calculation Strategy

Previous long-term environmental assessments at the Hanford Site have consistently shown that the groundwater pathway is the most important. This pathway also requires the most calculations. The conceptual model used for this and earlier Hanford Site performance assessments take the following eight steps (see Figure 3-2):

1. The water leaves the very-near-surface soil region at the infiltration rate.
2. The water moves toward the waste form, but most of it is diverted by any intact capillary barrier.
3. The water that is not diverted is chemically modified by the local environment, interacts with the waste form, accumulates contaminants, and again is chemically modified by the local environment.
4. The water (possibly a reduced amount) leaves the disposal facility carrying contaminants with it. Some contaminants may interact with the material in the disposal facility, slowing the release of the contaminants to the surrounding natural environment.
5. The water moves through the undisturbed, unsaturated zone (vadose zone) below the disposal facility down to the unconfined aquifer. The contaminants also are transported through the vadose zone, again possibly undergoing some geochemical sorption.
6. The water and contaminants move and mix with the water in the unconfined aquifer until they are extracted from the aquifer and brought to the surface or until they reach the Columbia River.
7. Contaminants are normally extracted by being carried to the surface with groundwater being pumped through a well.
8. The radionuclide contaminants then result in human exposure through a variety of pathways (ingestion, inhalation, and external radiation).

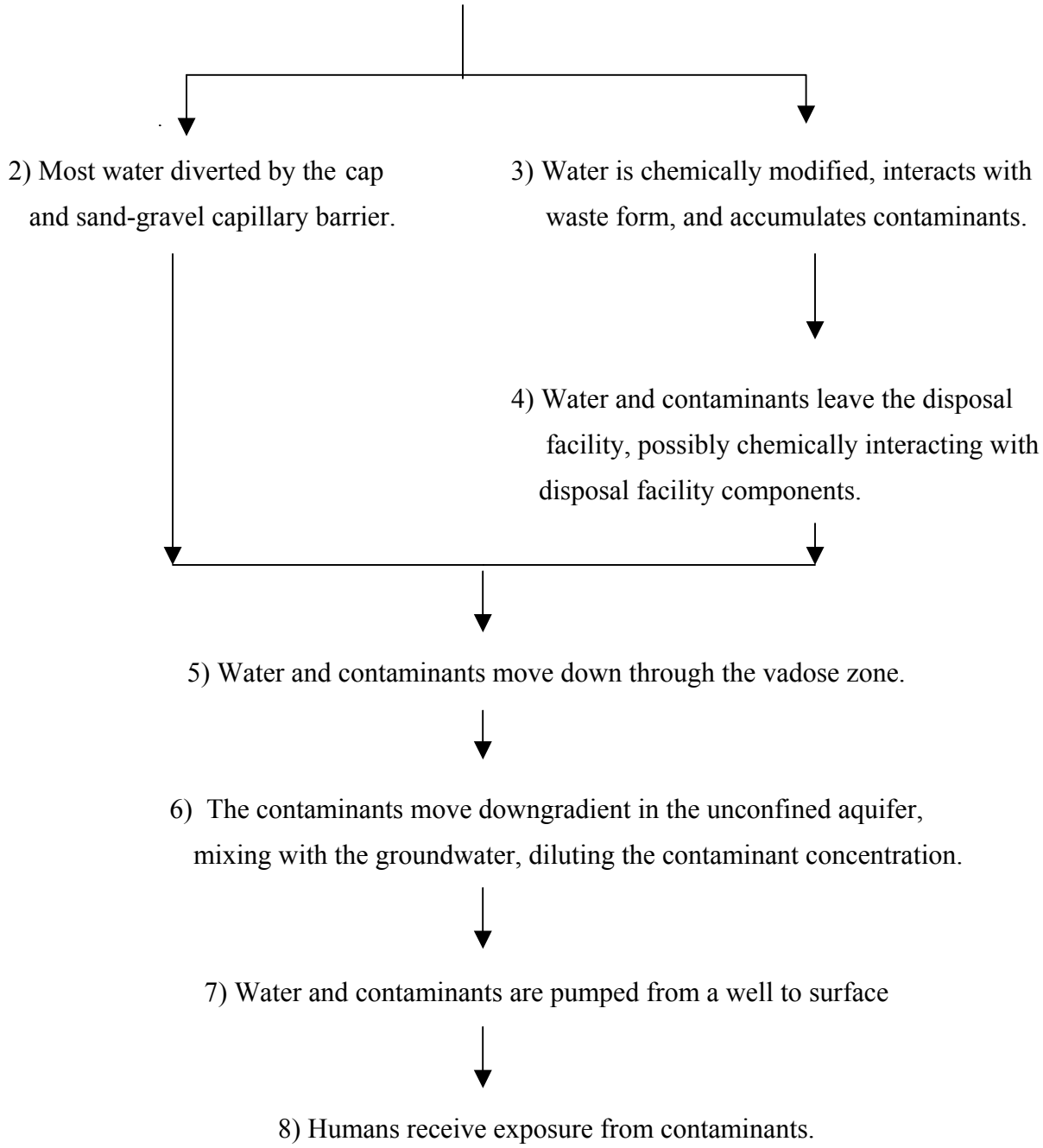
The results for each step are computed separately and used in the next step so that computations can be made more easily. Such an approach is taken to maximize computational efficiency. The overall model is always considered at each step and consistent data are used throughout.

The strategy for the current computations is to define a reference case, then develop other selected cases derived from that reference case to provide a limited estimate of the sensitivity of the results to key parameters. The different potential source terms within the IDF (ILAW LLW/MLLW and melters) are treated separately and then later combined to estimate the integrated risk from the proposed disposal action.

Computer codes have been used for four purposes:

- to calculate contaminant release rates from the ILAW waste packages and from the disposal facility (Bacon and McGrail 2002),
- to calculate moisture flow and contaminant transport in the vadose zone (including moisture flow into the disposal facility),
- to calculate moisture flow and contaminant transport in groundwater, and

Figure 3-2. Eight Sequential Steps for the Groundwater Pathway.



- to merge the results of the preceding codes.

Figure 3-3 illustrates also the overall computational strategy for the IDF risk assessment.

3.4.1.1 Numerical Model. Figure 3-4 shows the conceptual model used for the numerical calculations performed on this risk assessment. This conceptual model consists of a near-field zone and a far field zone. The half-trench model assumes symmetry about the X=0 vertical plane. The near-field model is used to choose the appropriate infiltration and contaminant flux profile for the different waste forms into the vadose zone that is represented by the far field model. The far field model is then incorporated into a computer model to calculate the contaminant flux into the unconfined aquifer.

Within the near field model the RCRA cap and capillary break are peaked along the north-south axis of the trench and has a 2% downward slope. The cap extends 10 meters beyond the end of the inner edge of the trench. Details of the trench geometry are defined in Section 2.3. The top of the IDF trench is modeled to be at Z=0 meters. The LCRS and the sloping bottom of the trench are neglected in the model calculations. The bottom of the near-field zone is chosen to be at Z=-17.8 meters. This depth was chosen to be consistent with the near-field contaminant release calculations for the ILAW waste form (Bacon and McGrail 2002), which included 5 m from the bottom of the lowest waste package to the bottom of their model. Details on the waste package distributions and corresponding contaminant release rate estimates for this risk assessment are defined in Section 3.4.3.

The far field model begins at the depth of Z=-17.8 meters and extends downward to the top of the unconfined aquifer. Based on the post-closure calculations of Bergeron and Wurstner (2000), this depth is estimated to be at Z=-103 meters. As in the 2001 ILAW PA, the far field is modeled with two material zones: the Hanford sands and the Hanford gravelly formation. The interface between these two materials is based on borehole information (see Reidel and Horton 1999 and Figure 2-2). The hydraulic and geochemistry parameters defining this region are provided in Sections 3.3.3.2 and 3.3.3.3, respectively.

3.4.1.2 Reference Case. The reference case for this risk assessment is based on the "best" information on how the system may evolve given the information available. The reference case is not necessarily the way the system will behave, but presents an idealized, conservative representation of how the system might perform. The approach used in the reference case is conservative, but reasonable.

The numerical model is based on the dimensions provided in Figure 3-4. The details of the models and related data for the reference case are presented in Sections 3.4.3 and 3.4.4, respectively. The major features of the reference case are as follows:

- The location of the facility is that selected for the new ILAW disposal facility (Rutherford 1997)
- The future land use of the 200 Areas is as a protected area, without artificial recharge (for example, no irrigated farming occurs)
- The design of the disposal facility is based on a 30% preliminary design for the IDF (Section 3.3.5)

Figure 3-3. Modeling Strategy for Assessing the IDF System.

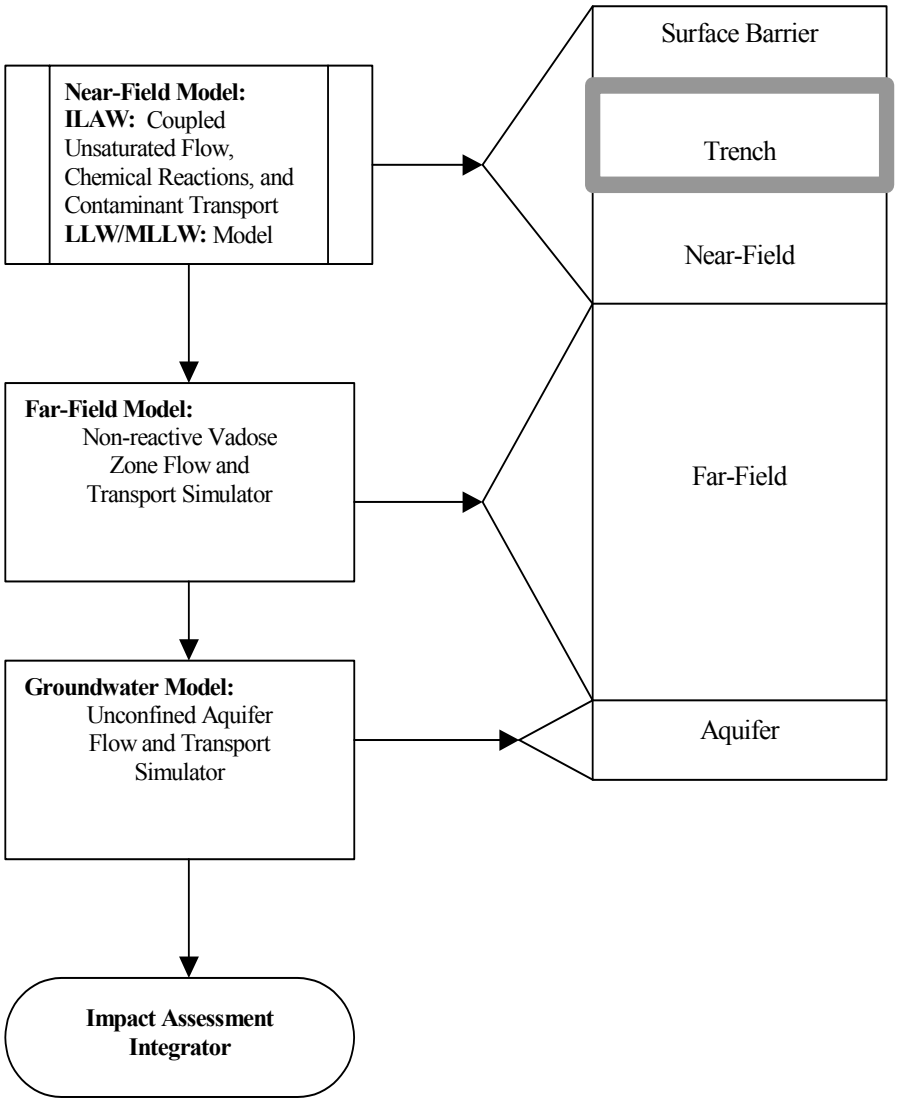
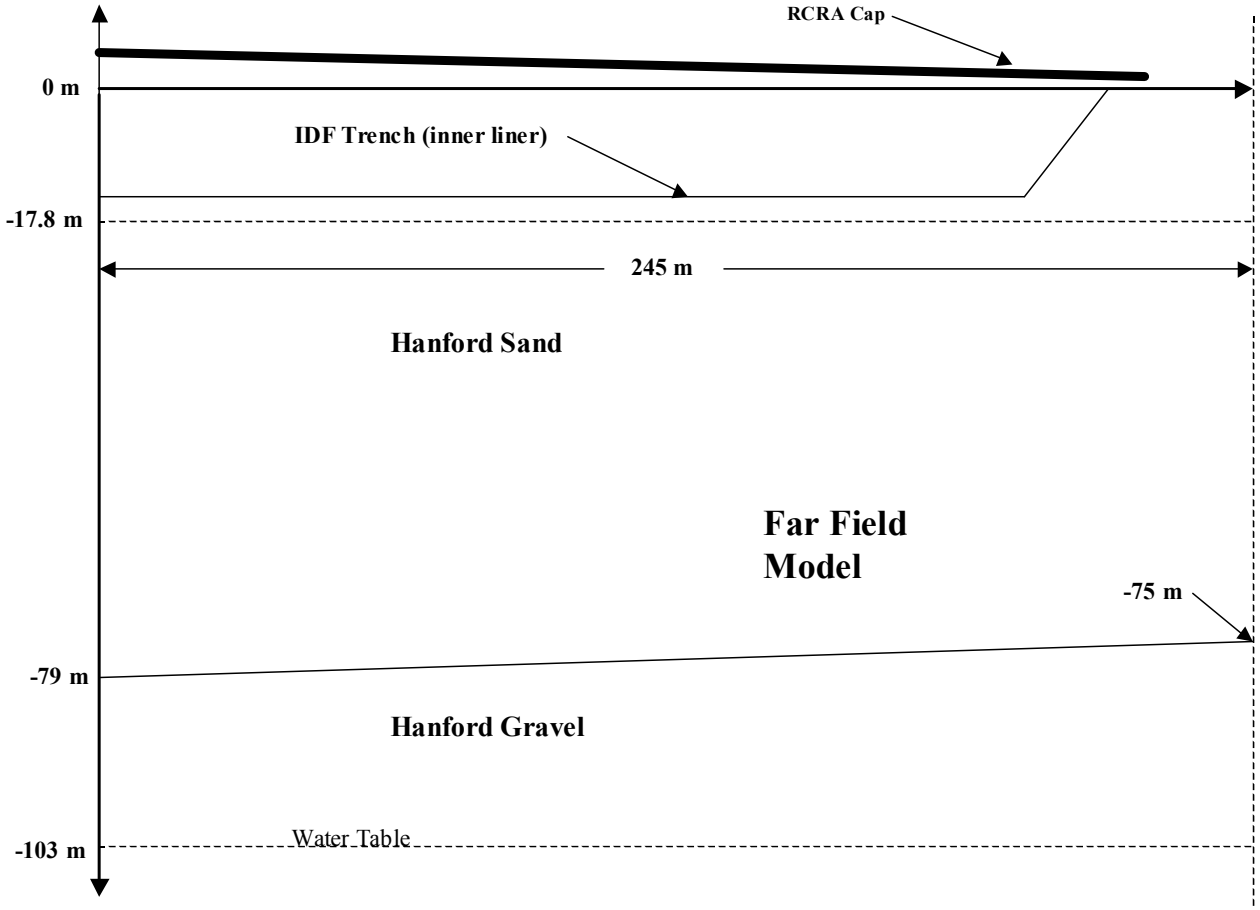


Figure 3-4. Conceptual Model for the IDF Trench.



- The long-term contaminant release rates from the different waste forms are calculated as described in Section 3.4.3.
- The data for the natural system are those collected and interpreted for the 2001 ILAW PA (Mann et al. 2001).

For the reference case we have assumed the surface barrier and capillary break are functional for the first 500 years after facility closure. The resulting infiltration rate into the trench is assumed to be 0.1 mm/y. This assumption is conservative because it assumes a higher infiltration rate than anticipated based on a functioning capillary break. A capillary break will lead to lower infiltration rates than 0.1 mm/y based on the calculations from the 2001 ILAW PA. The surface barrier and capillary break are assumed to fail 500 years after facility closure. The recharge rate for Burbank loamy sand is assumed for times greater than 500 years. The recharge rate for Burbank loamy sand was selected since it is the larger of the two potential recharge rates for that region and it was the recharge rate used for the ILAW base analysis case.

For the reference case we have also neglected the high recharge rate associated with the cover side slope region. Calculations in the 2001 ILAW PA (Mann et al. 2001) have shown that the effect of this high recharge is small at 10,000 years after facility closure (see Section 4.6.2, no side slope case, in Mann et al. 2001).

The major parameter in the reference case is the assumed recharge rate in the model. Separate reference cases are calculated for each waste form type (ILAW and LLW/MLLW). The results for each waste type are normalized to the quantity of waste in the model. The total performance of the system is then analyzed using superposition of the various sources to estimate the impacts to the environment for various exposure scenarios.

3.4.1.3 Other Cases. The other cases included in this risk assessment are a base case and other release models. These other cases use the same dimensional model shown in Figure 3-4. The base case differs from the reference case only in the assumed recharge rate. For the base case the recharge rate into the model is assumed to be 4.2-mm/y beginning at facility closure. The case is similar to the 2001 ILAW PA base analysis case and assumes both the RCRA cap and capillary break are not functioning as a moisture barrier.

The other cases investigated in this risk assessment are the upper bound inventory and a pulse / constant release model. The pulse / constant release model provides insights into the effect of different release rates on the contaminant fluxes into the unconfined aquifer.

3.4.2 Near Field Flow Calculations

For this IDF risk assessment we have assumed a conservative flow field at the top of the far field model to estimate the near field moisture flow through the facility. Specifically, for the reference case we have assumed that the RCRA cap and capillary break beneath the cap will function for only 500 years after facility closure. During this period the recharge rate through the region uniformly beneath the RCRA cap is assumed to be 0.1 mm/y (based of Fayer [1999] estimate for recharge rate beneath the RCRA cap and assuming capillary break beneath the RCRA cap does not function). The recharge rate into the facility after 500 years is assumed to be equal to the natural recharge at the site. The recharge rate in the region beyond the RCRA cap is assumed to be the same as the recharge rate into the trench as a function of time. The lateral spread of the high side slope recharge into the waste-bearing region of the IDF has been neglected in this risk assessment. Based on the 2001 ILAW results, the neglect of this effect is anticipated to be small.

The 2001 ILAW PA near field flow calculations (Finfrock 2000) were performed using the VAM3DF code to determine the moisture flow into the remote handled trench facility. The near-field region (primarily the engineered facility) significantly alters the flow field from what would be present in an undisturbed environment if the RCRA cap and capillary break beneath the RCRA cap are functioning. The best estimate case assumed the capillary break beneath the RCRA cap was functional for all time. The base analysis case assumed the recharge rate into the trench beneath the RCRA cap was 4.2 mm/y. Both these cases assumed the recharge rate beyond the edge of the RCRA cap was 50 mm/y.

The 2001 ILAW PA best estimate case near field calculations assumed a recharge rate of 4.2 mm/y above the capillary break and resulted in a maximum Darcy velocity within the trench of 8.5×10^{-3} mm/y (Finfrock 2000). For the IDF risk assessment we have assumed that the recharge rate into the trench is 0.1 mm/y for the first 500 years. This higher recharge rate is predicted to lead to somewhat faster contaminant transport times than would be expected assuming a functioning capillary break. Also, the release rates for the ILAW, and Category 1 LLW waste increase with increasing recharge rate. This higher recharge rate for the reference case will lead to higher contaminant releases from these waste forms. For these reasons the estimated impacts to the groundwater contaminant flux are anticipated to be initially higher than would be estimated using a more accurate model for the near field performance.

The 2001 ILAW PA base analysis case near field calculations assumed a recharge rate of 4.2 mm/y above the RCRA cap and a side slope recharge rate of 50 mm/y. The calculation resulted in a maximum Darcy velocity within the trench of 4.2 mm/y. A 2001 ILAW sensitivity case was run where the recharge rate in the side slope region was also 4.2 mm/y. The resulting estimated impact for the beta-photon dose at 10,000 years after facility closure was 3% less that the estimated impact for the base analysis case. At 1,000 years after facility closure, the difference in estimated impacts were larger (factor of 20); however, this difference is attributed primarily to the higher recharge rate in the side slope region.

3.4.3 Waste Form Calculations

For this risk assessment we have assumed the normalized waste form release for the different waste forms can be used as a source term at the top of the far field model to estimate the environmental risk. This approach is conservative because it neglects the transit time of the contaminants through the near field (estimated to be small). The approach for estimating the normalized waste form releases is described in the following sections for the ILAW waste form, the LLW/MLLW waste form, and the pulse release model.

3.4.3.1 ILAW Waste Form. The *Subsurface Transport Over Reactive Multiphases* (STORM) code (Bacon et al. 2000) is the source-term code used for estimating the time-dependent flux of radionuclides released from the ILAW waste form and the subsequent transport of contaminants in the disposal facility. STORM contains two important aspects that allow the code to simulate the processes in the disposal facility. First, the code is based on basic principles of physics, chemistry, and thermodynamics that provide the best estimate of contaminant release over the spatial and long time periods of interest. Second, the model for the disposal facility can be coupled with a model for radionuclide release, thus providing the ability to couple the effects of facility design with waste form performance.

Using chemical reaction rates (including the glass corrosion rates) and moisture content estimates in the trench, STORM provides the source term for the vadose zone calculations. STORM calculates the following:

- The flow of moisture in the disposal facility
- The degradation of the waste form with corresponding release of radionuclides
- The chemical reactions depend on time and space (including the formation of secondary mineral phases and the consumption of water)
- The transport of the water and contaminants through the disposal facility.

These one-dimensional calculations for the ILAW waste package design are reported in Bacon and McGrail (2002).

3.4.3.2 LLW/MLLW Waste Form. The waste form release models developed for the Solid Waste Burial grounds (Wood et al. 1995a and 1996) are used to estimate the release of contaminants from the LLW and MLLW planned for disposal in the IDF. Two model releases are explored for this risk assessment: 1) an advection model, and 2) a diffusion dominated release model.

The actual process of radionuclide release for the solid and mixed wastes cannot be modeled precisely because of the variability of chemical and physical reactions that occur in the waste material. In the real system, radionuclides are distributed in a heterogeneous manner, radionuclides are released into solution at different rates because of the variability in waste

material, and variable types and quantities of radionuclides are dissolved into the infiltrating water over time, depending on which waste material contacts a particular fluid volume. Therefore, averaging concepts are used in modeling to simplify the mathematical representation of the real system. These concepts must be justified, however, as being a conservative representation of the real system.

The following assumptions are made for the source-term release estimates.

- The release of contaminants is evaluated assuming that the recharge (infiltrating) water enters the facility, dissolves contaminants from the waste materials, and the release of contaminants occurs by dissolution of infiltrating water migrating into and out of the facility. It is assumed that advection-dominated models describe release of contaminants from the LLW Category 1materials. A diffusion-dominated model is assumed for LLW Category 3 and MLLW materials.
- For both stabilized (grouted) and unstabilized (i.e., not grouted) wastes in the facility, it is assumed that the contaminant inventory will be immediately available for release into the infiltrating solution.
- Unit quantities will be assumed for the modeling runs. Because dose estimates are directly proportional to initial inventory, the modeling runs with unit quantities can be scaled to calculate dose for any initial inventory values. (Note, we have conservatively assumed that there are no solubility limits involved. See Section 4.1.1.5 for additional discussion on solubility limits.)

The mathematical description and conditions under which the different mechanisms occur are provided in the following sections. The integral over time for both release models equals to unit inventory.

3.4.3.2.1 Advection-Dominated Release Model. The advection-dominated release model (mixing-cell cascade model) is used to simulate the release from the LLW (Category 1) materials. The radionuclides exit the facility at a rate determined by the flow of water and the amount of dispersion (mixing) within the facility. The mixing-cell cascade model (Kozak et al. 1990) is based on the dispersion analysis of chemical reactors and allows the analysis to incorporate the effects of dispersion in the facility in a simplified manner. In this model, the waste material inside is considered to be composed of a cascade of N equal-sized, well-stirred cells in series. The total volume of the N cells is equal to the volume of the waste material.

The mixing-cell cascade model for N equal-sized cells is described by the following equation:

$$Q(t) = q\,A\,C_o\,e^{-\alpha Nt}\sum_{n=1}^{N}\frac{(\alpha Nt)^{n-1}}{(n-1)!} \qquad (3.3)$$

where:

Q = release rate (Ci/y)

q = vertical Darcy flux (m/y)

A = horizontal (planar) area of the facility

$\alpha = q/(\theta dR)$

θ = volumetric moisture content in the waste material

d = vertical depth of the waste material (m)

R = retardation factor in the waste material (assumed R=1).

The initial concentration of contaminant in the interstitial water can be determined from the following equation:

$$C_0 = \frac{m}{\theta VR} \tag{3.4}$$

where m equals total facility inventory (assumed unity) of the radionuclides in the facility and V equals total volume of the waste material. A θ value of 10% is used (Wood et al. 1995a).

The mixing-cell cascade model provides results equivalent to the one-dimensional, convective-dispersion equation with varying values of the dispersion coefficient (Kozak et al. 1990). In the limit, as N approaches infinity, the model represents flow through a system with zero dispersion, whereas for N equal to one, the model represents flow with an infinite dispersion coefficient. A value of N = 10 is used reflecting moderate dispersion.

3.4.3.2.2 Diffusion-Dominated Release Model.

The diffusion-dominated release model is used to simulate the release of contaminants from stabilized (e.g., grouted) wastes (LLW Category 3 and MLLW). The diffusion from cylindrical containers leads to an expression for flux that contains infinite series (Kozak et al. 1990). The series converges slowly for small diffusion coefficients for short times, and even for relatively long times. As a result, the solution for a one-dimensional diffusion equation can be adopted (Crank 1975). The solution, for a semi-infinite medium with the concentration C_0 throughout, initially, and with zero surface concentration, is given by

where:

$$C = C_0 \, erf \frac{x}{2\sqrt{(D_e t)}} \tag{3.5}$$

erf = standard error function,

D_e = effective diffusion coefficient of the radionuclides in the waste form, and

t = time.

The rate of loss of diffusing substance per unit area from the semi-infinite medium when the surface concentration is zero, is given by:

$$\left(D_e \frac{\partial C}{\partial x}\right)_{x=0} = C_0 \sqrt{\frac{D_e}{\pi t}} \tag{3.6}$$

The above equation has the form of diffusive mass transfer based on leaching theory. This simplified release model leads to the following form:

where:

q = release rate from a single waste cell (Ci/y),

$$q = A\, C_0 \sqrt{\frac{D_e}{\pi t}} \tag{3.7}$$

A = effective surface area of a single cell, and

C_0 = contaminant concentration in a cell.

Because the waste material is likely contained in various cells with differing sizes and shapes, the diffusive release rate, Q, from all waste material in the facility can be determined by the following equation:

$$\begin{aligned} Q &= C_0 \sqrt{\frac{D_e}{\pi t}} \sum_{i=1}^{n} A_i \\ &= C_0 A_t \sqrt{\frac{D_e}{\pi t}} \end{aligned} \tag{3.8}$$

where n is the number of cells, A_i is the surface area of individual cells and A_t is total surface area of the facility.

By assuming that the cells are constant, i.e.,

$$I = C_0 \sum_{i=1}^{n} V_i = C_0 V_t \tag{3.9}$$

where I is total inventory, V_i is the volume of i-th cell and V_t is total volume of all cells.

Combining preceding equations, we obtain:

$$Q = I \frac{A_t}{V_t} \sqrt{\frac{D_e}{\pi t}} \tag{3.10}$$

The ratio A_t/V_t can be replaced by a ratio of a surface area over volume of the facility (only the portion of the facility containing waste is used to obtain the ratio).

The model calculation is conservative in two aspects. First, the surface area of the facility will not be completely exposed to a moving stream of water. Second, the radionuclides reaching the surface area are assumed to be released into the water stream and instantaneously reach the bottom of the facility. Recent disposal condition specific tests of Tc-99 and I-129 diffusion through grout in an unsaturated soil (Mattigod et al.2001) yield low diffusion coefficients (D_e) of 10^{-12} cm^2/sec or less. For this risk assessment we have assumed the contaminant diffusion coefficient of $1x10^{-11}$ cm^2/s for the reference case. (Note from the diffusion equation, the release rate is proportional to the square root of the diffusion coefficient. Therefore, increasing the diffusion coefficient by a factor of 10 increases the estimated release rate by approximately a factor of 3.2.)

3.4.3.3 Pulse/Constant Release Waste Form Model. Finally, a pulse/constant release model is considered as a special release case where the entire inventory within the IDF is assumed to be released at a constant rate over a range of fixed periods (1 y, 100 y, 1000 y, and 10000 y) starting immediately after facility closure.

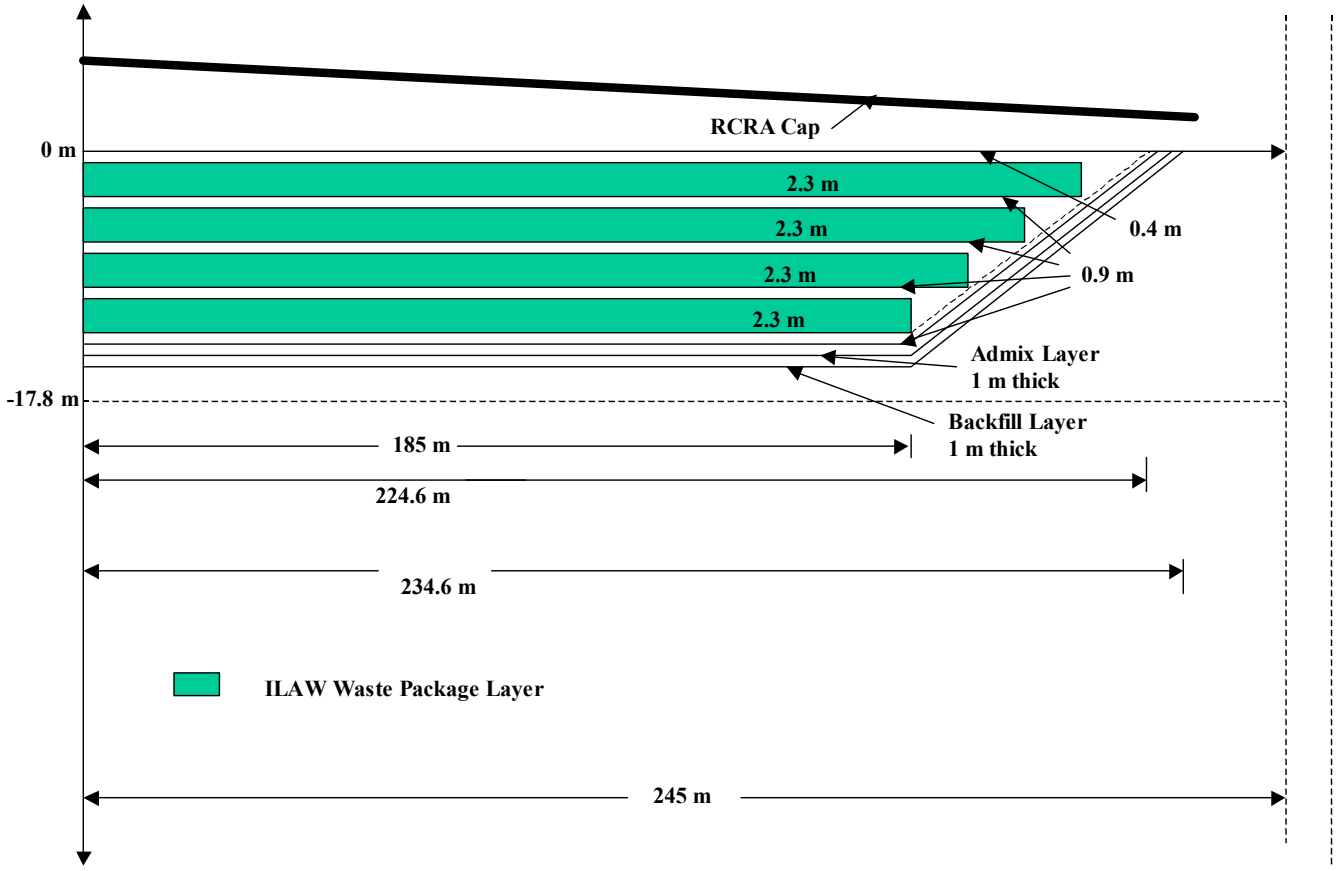
3.4.3.4 Geometrical Distribution of Waste Form in Model. Figure 3-5 shows the maximum distribution of ILAW waste packages within the IDF trench. As seen in the figure the ILAW waste packages can be stacked in 4 layers within the IDF trench with 0.9 meters of backfill between each layer.

For the LLW/MLLW waste form and the pulse/constant release model, the release is calculated as a source term boundary condition at the top of the far field model. The release is assumed to start at facility closure. This approach provides a conservative estimate of the contaminant flux entering the far field model because it neglects the contaminant transport time within the near field model zone. The normalized source term used in the numerical calculations assumes the same release rate as a function of time over the area bounded by 9.4 meters in height within the region above the bottom of the trench (out to x = 185 m, see Figure 3-5). On the side slope of the model the waste height decreases to 0 along the 3:1 slope. For the LLW/MLLW and pulse/constant release model we have assumed a uniform distribution of waste in a region bound by Z= 0 and the sloped line of the trench liner.

3.4.4 Far-Field Calculations

The far field calculations were performed using the computer code VAM3DF (Huyakon and Panday 1999) to calculate the moisture flow and the contaminant transport. The far field is the region of the model domain that extends from the base of the engineered facility down to the top of the water table (see Figure 3-4). The far field modeling process simultaneously solves the flow and transport equations. The model is run for the first 1,000 years prior to facility closure assuming a constant infiltration rate of 4.2 mm/y. This initial calculation is performed to allow the flow model to reach equilibrium. The release from the facility is assumed to begin at facility closure. For the far field calculations closure was assumed to occur at 2030. The difference

Figure 3-5. IDF Near-Field Model Cross-Section Showing ILAW Waste Layers.



between this closure date and the anticipated closure at 2046 has a negligible effect on the estimated impacts.

3.4.5 Groundwater Calculations

A local model of the Hanford Site-wide Groundwater Model was developed for the ILAW disposal site and calculations were performed for post-Hanford conditions (Bergeron and Wurstner 2000). The well intercept factors (WIFs) developed for the remote handled trench model used in the 2001 ILAW PA (Mann et al. 2001) are scaled to apply to the IDF that has a larger facility footprint (see Section 3.3.7).

The case where the ILAW inventory is maximized in one portion of the IDF trench was analyzed using the WIF from Table 3-10 and the estimated minimum area into which the ILAW inventory can be loaded into the IDF trench (see Section 3.3.5).

3.4.6 Impact Assessment Integrator

INTEG (Mann 1996) is a computer code developed to take a time variant, normalized contaminant release rate from a waste facility to the groundwater, and determine the resulting expected dose rate.

The code first determines the concentration of each contaminant at the point of interest by solving the equation:

$$C_i(t) = I_i(t)*G_i(t)*WIF/[r*A] \quad (3.11)$$

where:

$C_i(t)$ = Concentration at the point of interest for contaminant i at time t

$I_i(t)$ = Inventory of contaminant i at time t

$G_i(t)$ = The normalized contaminant flux rate into the groundwater at time t for contaminant i

WIF = The well intercept factor (dilution factor from the release point to the exposure point)

r = Surface recharge rate

A = Surface area of facility

Subsequently, INTEG calculates the expected dose at the point of interest by solving the equation:

$$D(t) = \Sigma_i \ Df_i * C_i(t) \quad (3.12)$$

where i varies over all of the nuclides of interest and Df_i is the dose factor for nuclide i for a given exposure scenario.

The dose factors (including ILCR and HI), inventory, recharge rate, and surface area are all defined by the problem specification. The normalized contaminant flux rate into the unconfined aquifer is a result of the VAM3DF transport calculations where the appropriate flux rate is chosen based on the isothermal sorption coefficient (K_d) of the given nuclide (which is part of the problem specification). Finally, the time dependent contaminant inventory is the initial inventory (part of the problem specification) reduced by INTEG according to the given contaminant's decay rate (other nuclides may build up due to the decay of parent nuclides).

The recharge rate, facility area, and well intercept factors are all specified with a script file that control the execution of the code. This file also specifies the names of files that contain the remainder of the input data. The initial inventory, nuclide decay factors and K_d's, and dose factors are all specified with library files. The primary results of the INTEG calculations are time dependent doses (drinking water and all-pathway) for a given exposure scenario and alpha emitter concentration. These data are written to an output file. Similar data are written to another file for each nuclide in the database.

3.4.7 Superposition of Different Waste Type Sources

To estimate the impacts to the groundwater from the IDF we have assumed the inventories for the different waste forms are averaged over the minimum trench volume needed to contain the estimated inventories. The estimated impacts for the various scenarios and groundwater concentration estimates are based on the superposition of the contributions from the different waste forms.

3.4.8 Input Data

This section specifies the data actually used in the computer models for the base analysis case. The intent is to follow the data given in Sections 3.1, 3.2, and 3.3 as closely as possible.

3.4.8.1 Contaminant Release Data. The data for the calculation of contaminant release rate from the different waste package types are those given in Sections 3.4.3. Table 3-14 summarizes the important input data.

3.4.8.2 Vadose Zone Data. The input data used for this risk assessment are those given in Section 3.3.3 and are summarized in Table 3-14.

3.4.8.3 Aquifer Modeling. The WIFs used for this risk assessment are based on scaling the 2001 ILAW PA aquifer modeling and are provided in Table 3-10.

3.4.9 Integration of Results.

In addition to data already discussed, the input data for INTEG were taken from the output of the vadose zone and the aquifer model. Inventory estimates were provided in Section 3.1.2. Dose conversion factors were provided in Section 3.3.5. The integrated results were estimated as the sum of the contributions from ILAW, LLW (Category 1) and LLW (Category 3)/MLLW inventories.

Table 3-14. Reference Case Input Data for the Integrated Disposal Facility.

Parameter	Value	Section with Justification for Using Value
Stratigraphic Cross-Section (two dimensional model used)		
Hanford formation: Upper Gravel Sequence Sand Sequence Lower Gravel Sequence	(nominal layer thickness) 7 m (23 ft) (on surface) 72 m (236 ft) 31 m (102 ft) (bottom)	Sections 2.1.3 and 3.3.3.2
Hydrologic Parameters		
Vadose Zone Soils	Calculated based on curve-fitting parameters and saturated hydraulic conductivity. See reference section.	Section 3.3.3.3 (Values given in Table 3-6)
Infiltration Rate		
At the Disposal Facility 500 years after facility closure >500 years	 0.1 mm/y 4.2 mm/y	Section 3.3.7

Table 3-14. Reference Case Input Data for the Integrated Disposal Facility.

Parameter	Value	Section with Justification for Using Value
Geochemical Parameters		
Chemical Distribution Coefficients (K_d) Tc, I, NO_3, Cr^{6+}, Others U, Np, Pa, Ru Se, C Sr, Ra Cs, Nb, Ni, Pb, Sn Ac, Am, Ce, Cm, Co, Eu, Pu, Th, Zr	 0.0 mL/g 0.6 mL/g 4.0 mL/g 10.0 mL/g 80.0 mL/g 150.0 mL/g	Section 3.3.3.4
Inventory		
ILAW	Based on 2001 ILAW PA. See reference Wootan 1999	Section 3.1.1
LLW/MLLW	Based on 2000 SWIFT report (Barcot 2002) and SW PEIS (DOE/EIS-0286D 2003)	Section 3.1.2 and 3.1.3
Melters	Based on BBI Tank Inventory and ILAW inventory (Wootan 1999)	Section 3.1.4
Contaminant Release Rate		
ILAW Radionuclide Release Rate	Calculated release based on model calculation. See reference Bacon et al. 2002	Section 3.4.3.1
LLW/MLLW Radionuclide Release Rate	Time dependent release model. See reference Wood et al. 1995a	Section 3.4.3.2
Well Intercept Factors		

Table 3-14. Reference Case Input Data for the Integrated Disposal Facility.

Parameter	Value	Section with Justification for Using Value
At different downgradient distances 100 m 1000 m Columbia River	Based on 2001 ILAW PA. See reference Bergeron and Wurstner 2000. 1.2×10^{-3} 6.6×10^{-4} 1.2×10^{-4}	Section 3.3.8

4.0 RESULTS OF ANALYSES

This chapter presents the results from the analyses described in Chapter 3. It also discusses the data and methods affecting the results. The chapter focuses on an understanding of the suite of calculations performed.

4.1 GROUNDWATER SCENARIO RESULTS

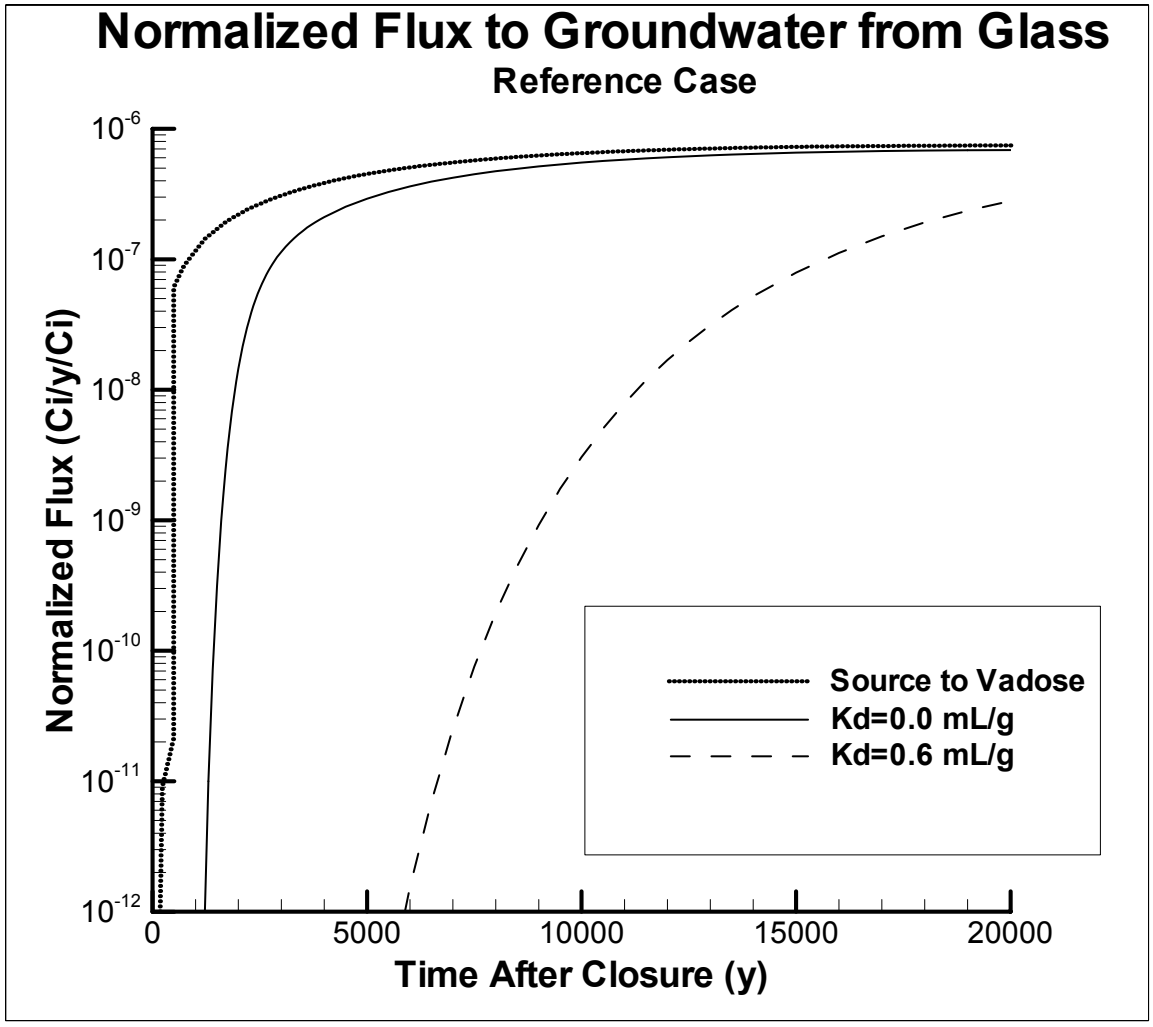
This section presents the results of the analyses for the reference case (Section 4.1.1) and for selected sensitive cases including the base case (Section 4.1.2.1), the bounding inventory case (Section 4.1.2.2), and various pulse/constant release cases (Section 4.1.2.3).

4.1.1 Reference Case Estimated Impacts

The reference case is based on an assumed recharge rate for the IDF trench. The RCRA barrier and capillary break are assumed to function effectively for 500 years after facility closure and limit the water infiltration rate into the facility to 0.1 mm/y (Fayer 1999). After 500 years the RCRA barrier and capillary break are assumed to no longer function and the infiltration rate into the trench is assumed to be equivalent to the vegetated Burbank loam infiltration rate of 4.2 mm/y (Fayer 1999) for all times greater than 500 years after facility closure. This section discusses the contributions to the estimated impacts from the different waste forms and provides estimated impacts compared to performance objectives for different exposure scenarios.

4.1.1.1 ILAW (Glass). Figure 4-1 shows the estimated contaminant flux at the top of the vadose zone (source to vadose), and for K_d = 0 and 0.6 mL/g at the bottom of the vadose zone normalized to the inventory for the ILAW glass waste form. The normalized contaminant flux is shown at Z=-17.8 m corresponding to the top of the far field model (source to vadose zone) and at Z=-103 m corresponding to the interface of the vadose zone with the unconfined aquifer (K_d = 0.0, 0.6 mL/g). For the contaminant flux at the top of the far field model, the shoulder in Figure 4-1 at times less than 500 years is associated with the lower contaminant release rate from the waste form due to the low recharge rate. At 500 years after facility closure the higher contaminant release rate associated with the 4.2 mm/y recharge rate is assumed. The estimated contaminant flux rate into the unconfined aquifer (Z=-103 m) for contaminants assigned a K_d = 0 mL/g is similar in shape to the waste form release rate and delayed approximately 2,000 years. The contaminants assigned a K_d = 0.6 mL/g do not have a significant flux to the unconfined aquifer until times greater than 10,000 years after facility closure.

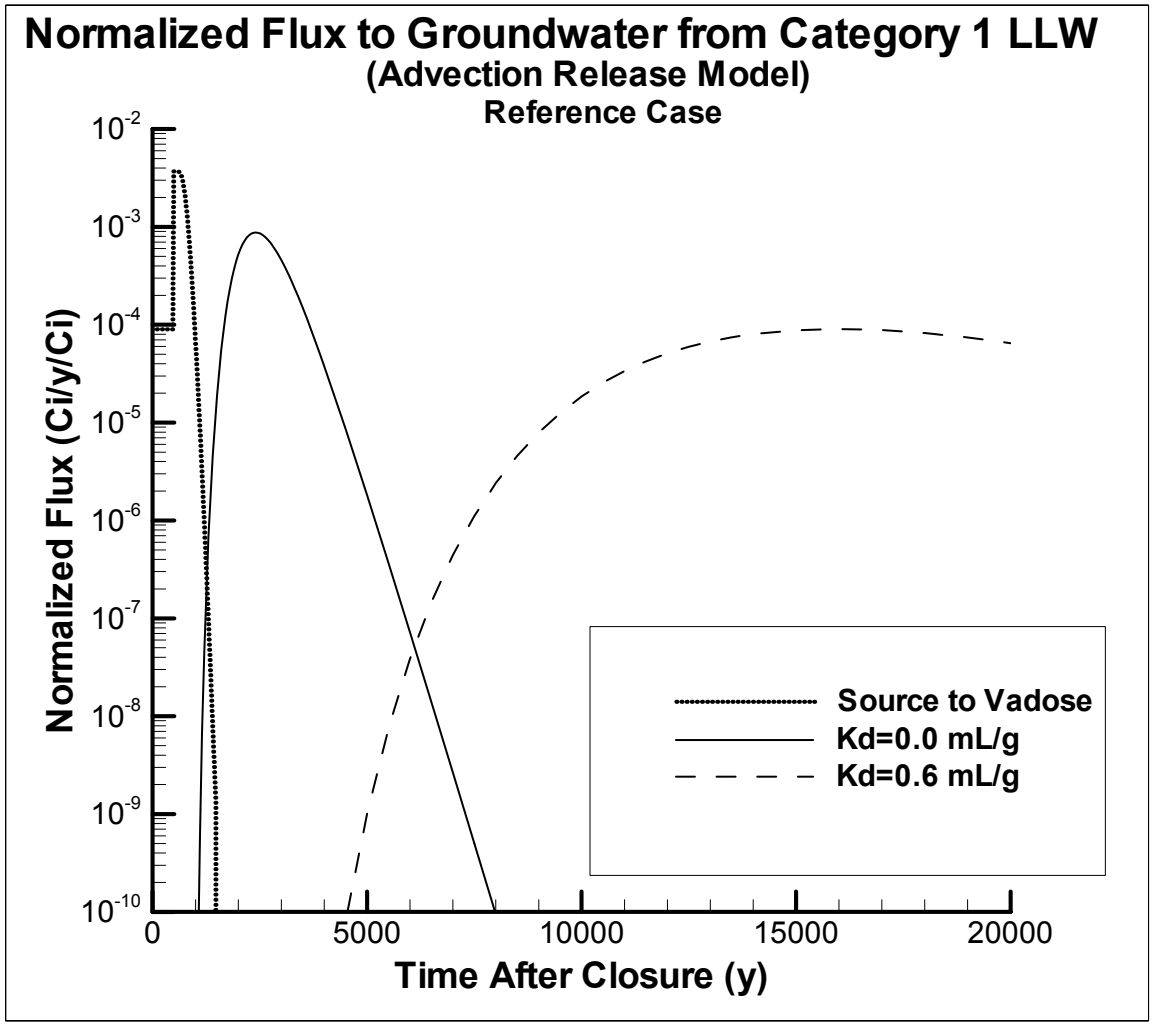
Figure 4-1. ILAW Contaminant Flux for the Reference Case.



4.1.1.2 Low-Level Waste (Category 1). Figure 4-2 shows the estimated contaminant flux at the top of the vadose zone (source to vadose), and for K_d = 0 and 0.6 mL/g at the bottom of the vadose zone normalized to inventory for LLW Category 1 waste. The contaminant flux is shown at Z=-17.8 m corresponding to the top of the far field model (source to vadose zone) and at Z=-103 m corresponding to a position at the interface of the vadose zone with the unconfined aquifer (K_d = 0.0, 0.6 mL/g). For LLW Category 1 waste, the waste form release is modeled with an advection-dominated release. For the contaminant flux at the top of the far field model, the approximate constant flux at times less than 500 years is associated with the lower contaminant release rate from the waste form due to the low recharge rate. For times greater than 500 years after facility closure the higher waste for release rate associated with the 4.2 mm/y recharge rate is assumed. This release rate is peak-shaped with a duration of approximately 300 years where the normalized flux is greater than ½ the peak estimated value. The estimated contaminant flux rate into the unconfined aquifer (Z=-103 m) for contaminants assigned a K_d = 0 mL/g is peak shaped with a full width at half maximum of approximately 1,000 years. The contaminant release rate at times less than 500 years has been smoothed out as

the contaminants are transported through the vadose zone. The normalized flux for contaminants assigned a K_d = 0.6 mL/g is approximately 25% of its maximum flux to the aquifer at 10,000 years after facility closure.

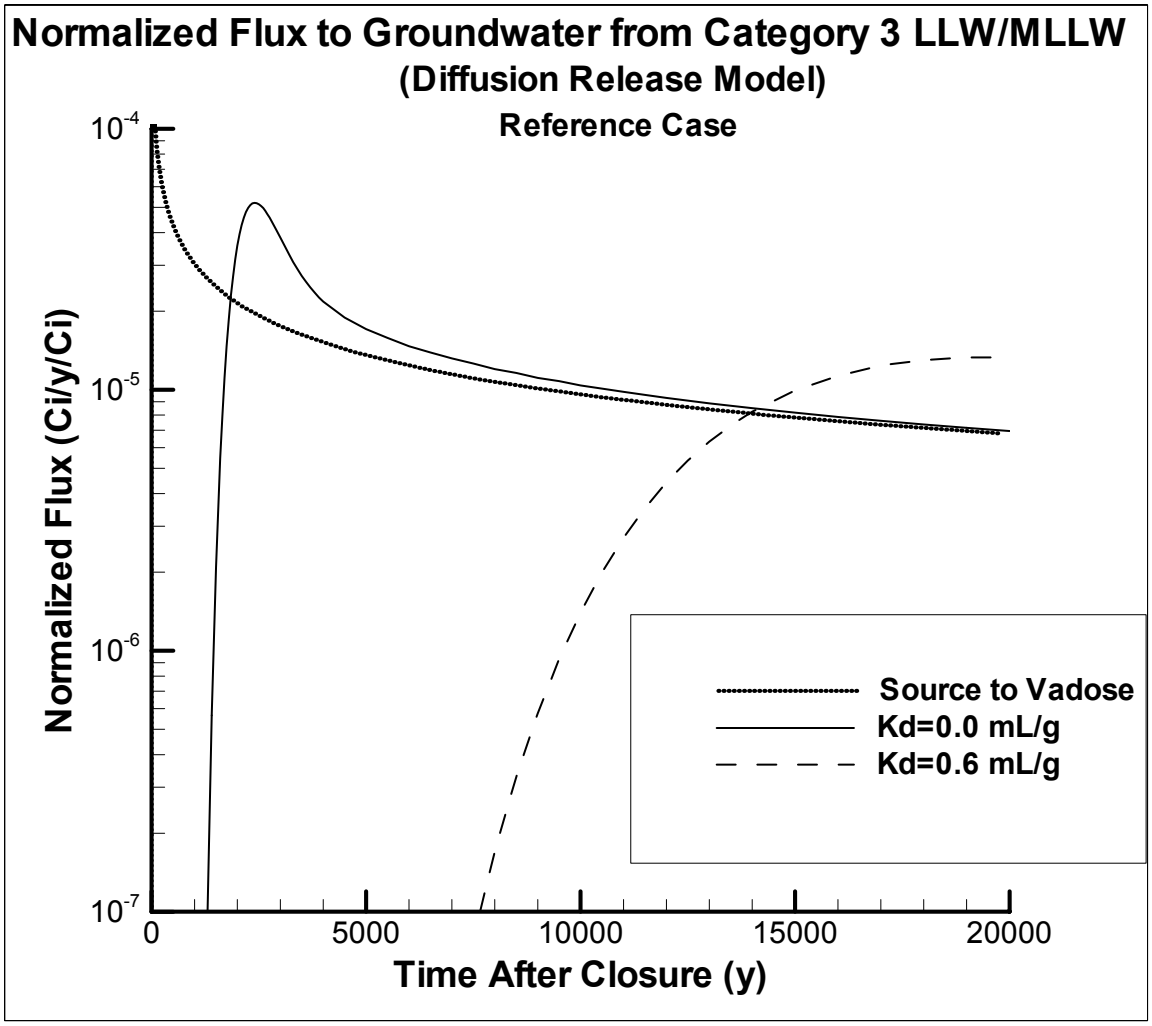
Figure 4-2. LLW (Category 1 – Advection Model) Contaminant Flux for the Reference Case.



4.1.1.3 Low-Level Waste (Category 3) / Mixed Low-Level Waste. Figure 4-3 shows the estimated contaminant flux at the top of the vadose zone (source to vadose), and for K_d = 0 and 0.6 mL/g at the bottom of the vadose zone normalized to inventory for the LLW Category 3 and MLLW waste form. For the LLW (Category 3) and MLLW waste form, the waste form release is modeled as a diffusion-dominated release with $D_e = 1 \times 10^{-11}$ cm^2/s for the reference case. The contaminant flux is shown at Z=-17.8 m corresponding to the top of the far field (Source to Vadose) and at Z=-103 m corresponding to a position at the interface of the vadose zone with the unconfined aquifer (K_d = 0.0, 0.6 mL/g). This release model is independent of recharge. This release rate is a strongly forward peaked release with a substantial contribution occurring at long times for the diffusion coefficient assumed in the model. The estimated contaminant flux rate

into the unconfined aquifer (Z=-103 m) for K_d = 0 mL/g is peaked at approximately 2400 years after facility closure and has a long "tail" with substantial contaminant fluxes out beyond 10,000 years after facility closure. The contaminants assigned a K_d = 0.6 mL/g do not have a significant flux to the unconfined aquifer until times greater than 10,000 years after facility closure.

Figure 4-3. LLW (Category 3) / MLLW – Diffusion Model Contaminant Flux for the Reference Case (D_e = 1 x 10^{-11} cm^2/s).



4.1.1.4 WTP Melters. Due to the relatively small inventory associated with the failed or decommissioned HLW and LAW melters, the environmental impacts due to their burial in the IDF trench are estimated to be small and have not been included in the total estimated environmental impacts of the IDF. Table 4-1 summarizes the inventory of the HLW and LAW melters compared to the ILAW and LLW/MLLW inventories for the mobile and moderately mobile contaminants of concern.

From Table 4-1 we find that the sum of the melter inventories for the contaminants of concern is approximately 6% of the inventory in the ILAW (except for Pa-231 that has a negligible contribution to the estimated doses based on the 2001 ILAW PA [Mann et al. 2001]) and is of the same order as the nominal inventory estimated for LLW and MLLW. Recall the

melter inventories are based on the assumption that all melters contain the maximum amount of glass-like waste form. Recall also that the waste form release rate from the melters is not known. This release rate may be comparable to the release rate measured for ILAW glass or may be as poor as the diffusion model release rate assumed for the LLW (Category 3) / MLLW waste form. Therefore, based on the inventory estimates and the contaminant fluxes estimated for the different release mechanisms described in Sections 4.1.1 through 4.1.3, the estimated environmental impacts from the melter disposal should be bounded by the estimated environmental impacts estimated for the LLW/MLLW waste form.

Table 4-1. Comparison of Contaminants of Concern Inventories.

Material	ILAW	LLW/MLLW	LAW Melter	HLW Melter	Sum Melters	Ratio to ILAW	Ratio to LLW/MLLW
Radionuclides (Inventory units are Ci)							
Tc-99	2.89E+04	1.10E+02	6.64E+01	0.00E+00	6.64E+01	2.30E-03	6.04E-01
I-129	2.20E+01	7.00E+00	5.05E-02	6.35E-01	6.86E-01	3.12E-02	9.79E-02
Pa-231	3.44E-01	0.00E+00	7.88E-04	1.25E+00	1.25E+00	3.64E+00	a
U-232	3.46E+01	0.00E+00	7.95E-02	9.20E-01	1.00E+00	2.89E-02	a
U-233	1.31E+02	0.00E+00	3.00E-01	3.55E+00	3.85E+00	2.94E-02	a
U-234	4.41E+01	2.00E+00	1.01E-01	2.40E+00	2.50E+00	5.67E-02	1.25E+00
U-235	1.79E+00	9.10E-02	4.10E-03	1.03E-01	1.07E-01	5.98E-02	1.18E+00
U-236	1.43E+00	0.00E+00	3.28E-03	8.82E-02	9.15E-02	6.40E-02	a
U-238	4.83E+01	2.00E+00	1.11E-01	2.25E+00	2.36E+00	4.89E-02	1.18E+00
Np-237	8.10E+01	5.00E+00	1.86E-01	8.36E-01	1.02E+00	1.26E-02	2.04E-01
Chemicals (Inventory units are g)							
Cl	9.31E+05	a	2.14E+03	4.83E+01	2.19E+03	2.35E-03	a
Cr (Total)	2.74E+05	a	6.28E+02	3.20E+03	3.83E+03	1.40E-02	a
NO_2	0.00E+00	a	0.00E+00	0.00E+00	0.00E+00	a	a
NO_3	0.00E+00	a	0.00E+00	0.00E+00	0.00E+00	a	a

a not estimated

4.1.1.5 Estimated Environmental Impacts. The estimated environmental impacts for the groundwater pathway are based on a superposition for the contaminant fluxes from the different waste forms and the dose factors associated with the different exposure scenarios. This section summarizes the results for the drinking water dose (associated with beta and photon emitter radionuclides and their associated decay daughters listed in the inventory tables [Tables 3-1 through 3-4]) all-pathways dose for the farmer scenario, alpha-emitting radionuclide concentration (associated with all radionuclides and their daughters that emit alpha particles

during its decay chain), radium concentration, uranium concentration, and the incremental lifetime cancer risk (ILCR) and hazard index (HI) associated with the contaminant concentrations estimated from the IDF (see Table 3-13).

Table 4-2 summarizes the estimated impacts from the reference case for a well 100 m downgradient from the disposal facility. Also shown in the table are the performance objectives for these estimated environmental impacts. The estimated impacts are given for 1,000 and 10,000 years after facility closure. The estimated impacts are for the maximum estimated impact within that time period. Figure 4-4 shows the relative contribution to the beta/photon dose from the different waste forms.

Table 4-2. Estimated Groundwater Impacts from the Reference Case for a well 100 m Downgradient from the Disposal Facility.

Type of Impact	Maximum Value within Time Period		Performance Objective
	1,000 years	10,000 years	
Dose (mrem in a year) from beta- and photon-emitting radionuclides in drinking water (Farmer Scenario [540 L/y])	4.7×10^{-11}	0.70	4.0
Concentration (pCi/L) of alpha-emitting radionuclides	0 [a]	0.19	15.0 [b]
Farmer Scenario - All-pathway dose (mrem in a year)	1.2×10^{-10}	1.8	25.0
Industrial Scenario - All-pathway dose (mrem in a year)	2.2×10^{-11}	0.32	25.0
Residential Scenario - All-pathway dose (mrem in a year)	7.3×10^{-11}	1.1	25.0
Radium Concentration (pCi/L)	0[a]	0[a]	3.0
Uranium Concentration (pCi/L)	0[a]	0.092	---

[a] The estimated impact is less than 1×10^{-20} pCi/L

[b] The performance objective excludes the uranium contribution to the concentration

Figure 4-4. Time Dependence of the Estimated Beta/Photon Drinking Water Dose at a Well 100 m Downgradient from the Disposal Facility.

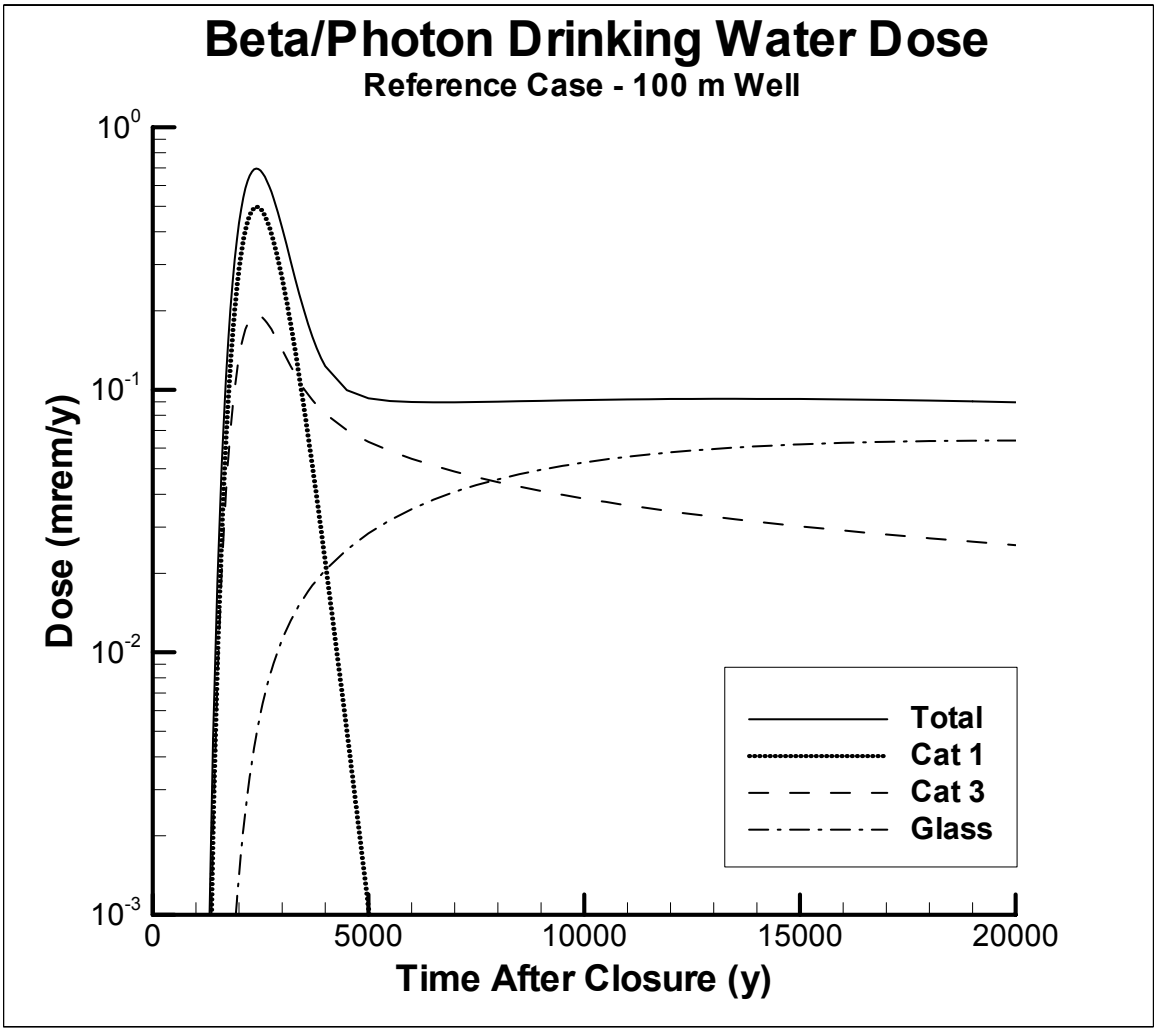


Figure 4-4 shows that the major contributor to the estimated beta/photon drinking water dose is the LLW (Category 1) inventory estimated for disposal into the IDF. Due to the initial low recharge associated with the reference case and the delay of the contaminants in reaching the unconfined aquifer due to transport through the vadose zone, the estimated impacts at times less than 1,000 years after facility closure are negligible. The peak dose is estimated to occur at approximately 2,400 years after facility closure. The estimated peak dose from the LLW (Category 1) waste is 0.50 mrem/y and the estimated peak dose from the LLW (Category 3) / MLLW is 0.20 mrem/y. (The contribution to the estimated dose at 2,400 years after facility closure from the ILAW glass is 0.005 mrem/y.) At 10,000 years after facility closure, the major contributors to the estimated dose are from the LLW (Category 3) / MLLW (42%) and ILAW (58%).

Figure 4-5 shows the alpha concentration in the groundwater at a well 100 m downgradient from the disposal facility for the reference case. The major contributors to the

alpha concentration in the groundwater are from the alpha emitters assigned a K_d = 0.6 mL/g (uranium and neptunium). Alpha concentrations greater than $1x10^{-3}$ pCi/L are not estimated to occur until times greater than 6,000 to 7,000 years after facility closure. The relative contributions from the LLW (Category 1) and the LLW (Category 3) / MLLW are 0.14 and 0.04 pCi/L, respectively, at 10,000 years after facility closure. The relative contribution from the ILAW is less than from the other two waste forms for all times out to 20,000 years after facility closure.

The estimated alpha concentration in the groundwater may be conservative due to solubility limits for uranium in its release associated with the advection and diffusion release models. The current estimates do not consider uranium solubility limits. At 10,000 years after facility closure the estimated uranium concentration in the pore water just above the aquifer is 0.02 mg/L. The U-238 concentration calculated by INTEG was divided by the WIF to give an average concentration at the bottom of the vadose zone model. A value of $3.36x10^{-7}$ Ci/g was used to convert the Ci concentration into a mass (g) concentration. The assumed solubility limits are 60 mg/L in soil and 0.24 mg/L in concrete/grout (Wood et al. 1995a). Application of a solubility limit to the release for the advection and diffusion release models may reduce the uranium concentrations entering the far field model and hence reduce the alpha concentrations in the groundwater.

Figure 4-6 shows the time dependence of the estimated all-pathways dose for the farmer scenario at a well 100 m downgradient from the disposal facility. The relative contributions from the three different waste forms are shown in the figure. Due to the initial low recharge associated with the reference case and the delay of the contaminants in reaching the unconfined aquifer due to transport through the vadose zone, the estimated impacts at times less than 1,000 years after facility closure are negligible. The peak dose is estimated to occur at approximately 2,400 years after facility closure. Table 4-3 shows the relative contributions to the farmer scenario all pathways dose at 1,000, 2,400, and 10,000 years after facility closure. The relative peak in the dose at 2,400 years after facility closure is due to comparable contributions from the K_d = 0 mL/g radionuclides (Tc-99 and I-129) contained in the LLW (Category 1) and LLW (Category 3) / MLLW. At 10,000 years after facility closure the radionuclides with K_d = 0.6 mL/g (U and Np) are also contributing to all-pathways dose. The relative contribution from the ILAW waste form is approximately two orders of magnitude less than the other waste forms at approximately 2,400 years after facility closure. At 10,000 years after facility closure all three waste forms contribute approximately equally to the all-pathway dose.

Figure 4-5. Time Dependence of the Estimated Alpha Emitter Concentration in the Groundwater at a Well 100 m Downgradient from the Disposal Facility.

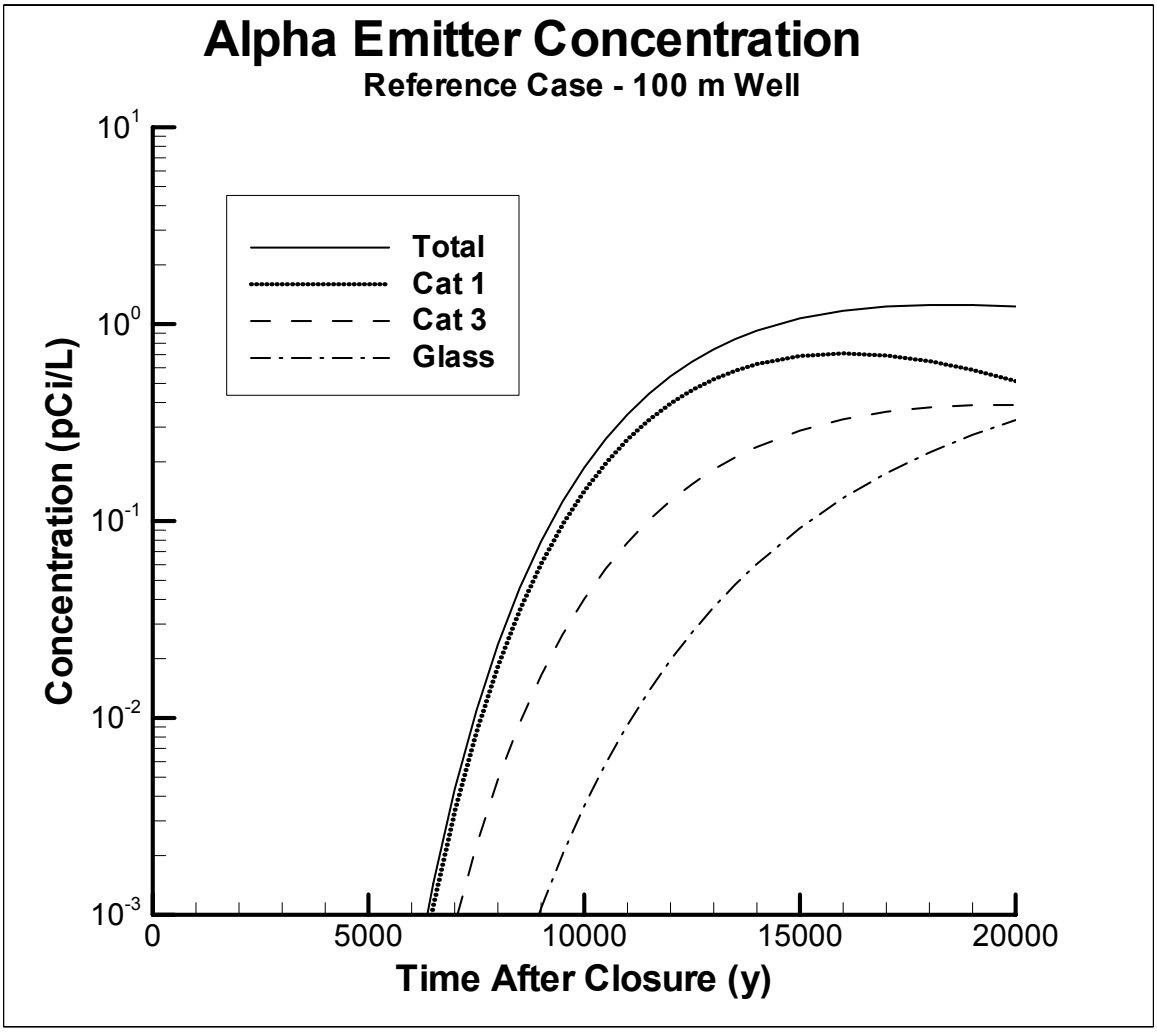


Figure 4-6. Time Dependence of the Estimated Farmer Scenario All-Pathways Dose at a Well 100 m Downgradient from the Disposal Facility.

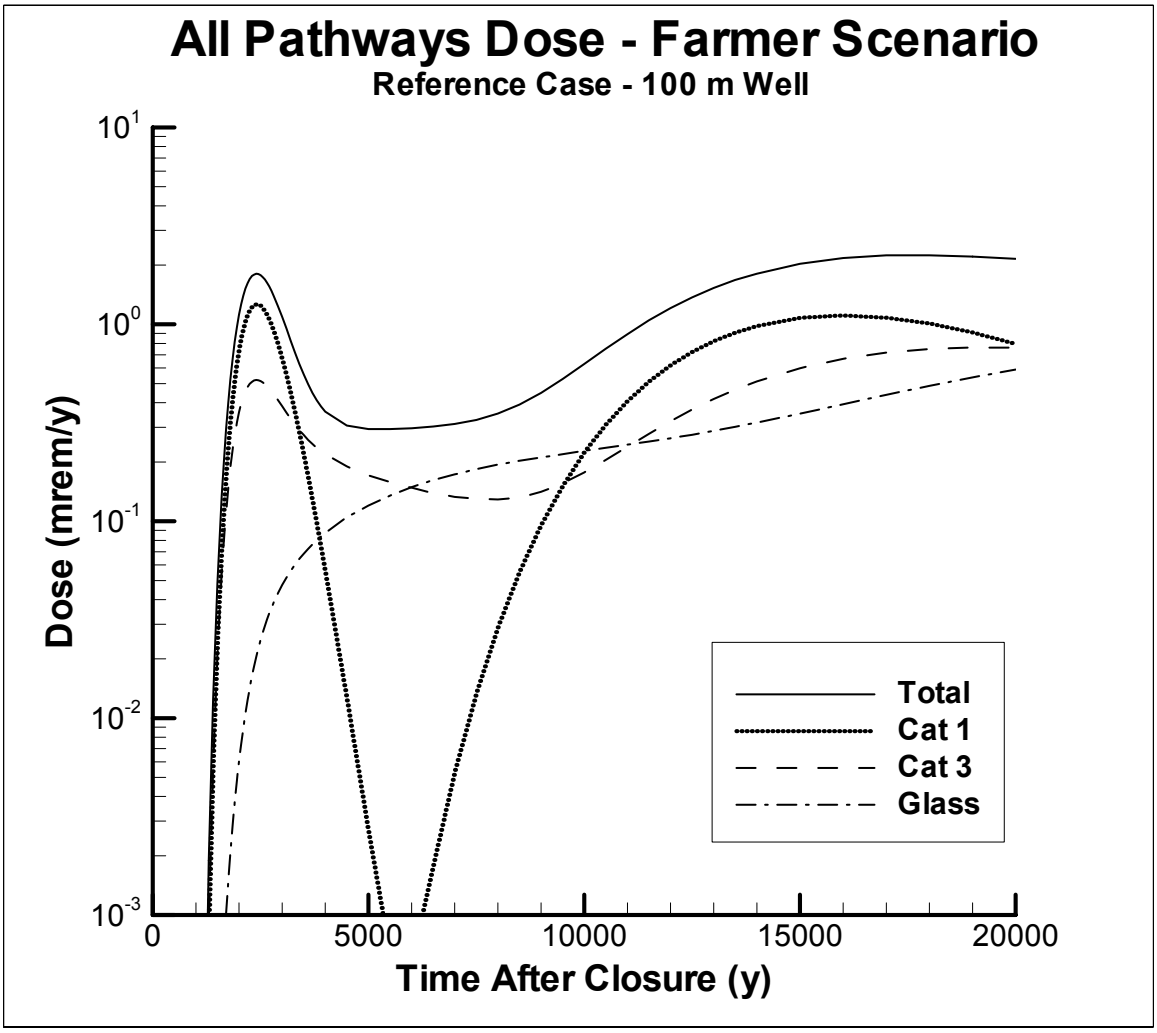


Table 4-3. Major Radionuclide Contributors at Various Times After Facility Closure to the Estimated Farmer Scenario All-Pathways Dose Due to a Well 100 m Downgradient from the Disposal Facility.

Radionuclide	1,000 y *		2,400 y *		10,000 y *	
	Dose (mrem/y)	Fraction	Dose (mrem/y)	Fraction	Dose (mrem/y)	Fraction
Tc-99	$1.1ex10^{-11}$	0.09	0.106	0.06	0.222	0.35
I-129	$1.13x10^{-10}$	0.91	1.70	0.94	0.107	0.17
Np-237	0	0	0	0	0.279	0.44
Other	0	0	0	0	0021	0.04
Total	$1.24x10^{-10}$		1.81		0.629	

* Time after facility closure

Figure 4-7 shows the time dependence of the estimated all-pathways dose for the farmer, industrial, and residential intruder at a well 100 m downgradient from the disposal facility. The results are similar to the farmer scenario all-pathway dose. Due to the initial low recharge associated with the reference case and the delay of the contaminants in reaching the unconfined aquifer due to transport through the vadose zone, the estimated impacts at times less than 1,000 years after facility closure are negligible. The peak dose is estimated to occur at approximately 2,400 years after facility closure. The relative peak in the dose at 2,400 years after facility closure is due to contributions from the K_d = 0 mL/g radionuclides (Tc-99 [6%] and I-129 [94%]) contained in the LLW (Category 1) and LLW (Category 3) / MLLW. At 10,000 years after facility closure the radionuclides with K_d = 0.6 mL/g are contributing the to all-pathways dose.

Figure 4-7. Time Dependence of the Estimated Farmer, Industrial and Residential Scenarios All-Pathways Dose at a Well 100 m Downgradient from the Disposal Facility.

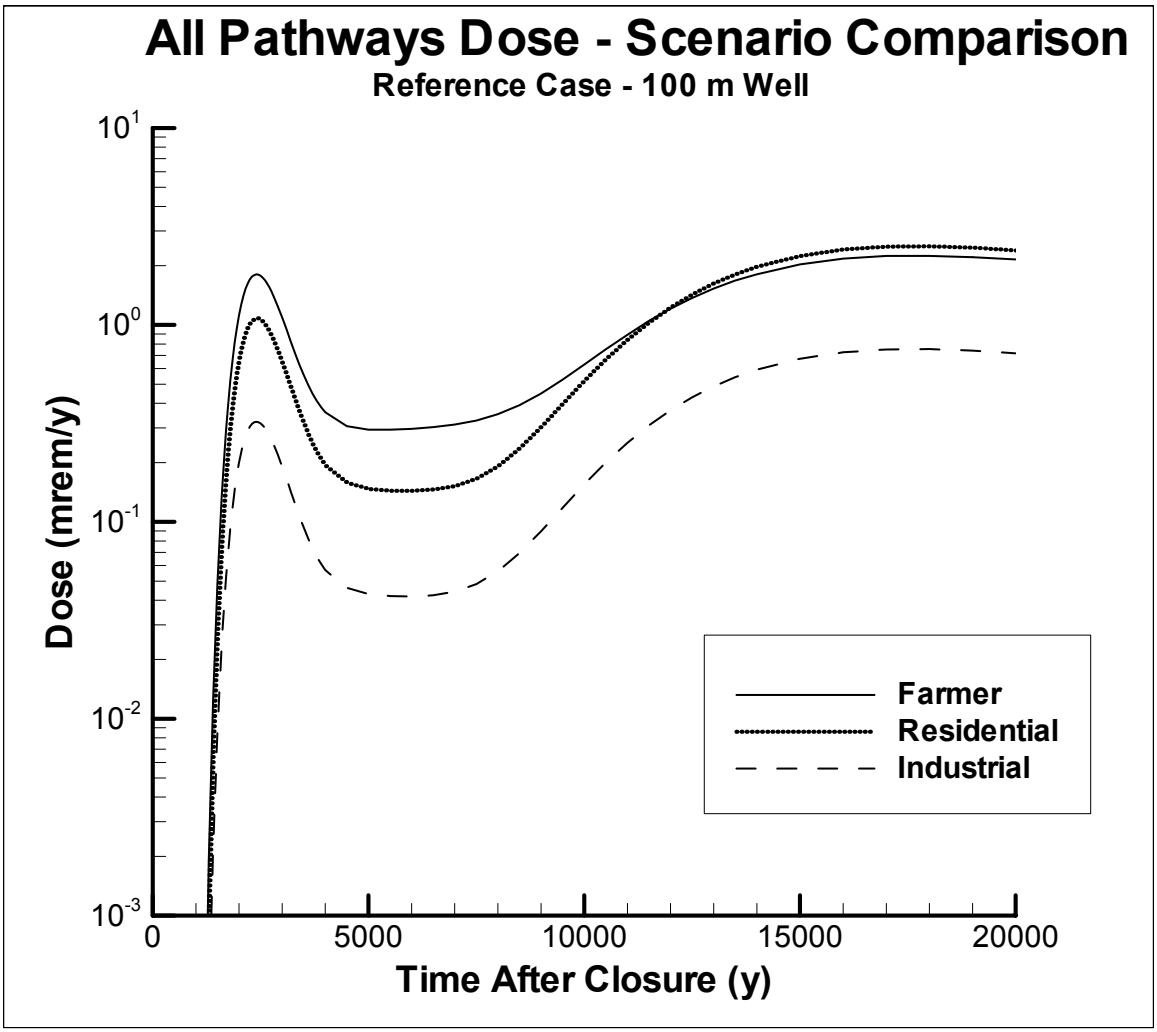


Table 4-4 summarizes the results for the all-pathway exposure at the Columbia River for the farmer and the Native American who resides near the Columbia River. The estimated doses

are based on the groundwater contaminant concentrations just before the groundwater enters the Columbia River. The estimated doses for the Native American are higher due to the definition of the Native American scenario.

Table 4-4. Estimated Impacts from the Reference Case at the Columbia River.

Type of Impact	Maximum Value within Time Period		Performance Objective
	1,000 years	10,000 years	
Farmer Scenario – All-Pathway Dose (mrem in a year)	6.1×10^{-12}	0.089	25.0
Native American Scenario - All-Pathway dose (mrem in a year)	2.54×10^{-11}	0.346	25.0

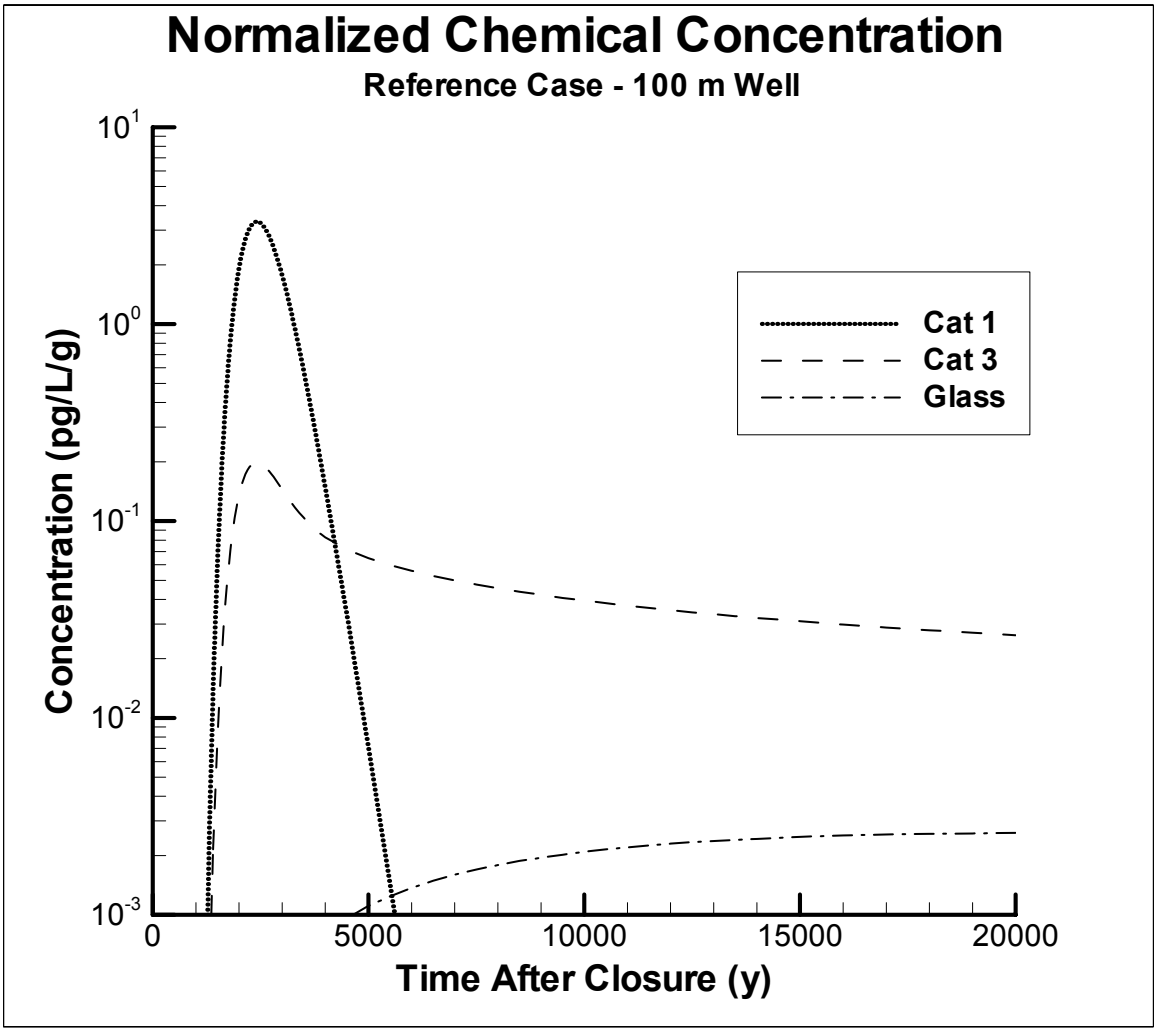
Finally, the impact from chemicals in the waste forms is small based on the inventory information we have. To assess the chemical impacts, the chemical concentrations in the groundwater at a well 100 m downgradient from the disposal facility were estimated. Because of our limited information on the chemical inventory for the LLW and MLLW, only the following chemicals were estimated: chromium, nitrate, and uranium. Table 4-5 compares the estimated chemical concentrations in the groundwater at a well 100 m downgradient from the disposal facility at various times after facility closure from the ILAW, LLW and MLLW. These concentrations are compared to performance goals documented in Mann (1999b). Figure 4-8 shows the contaminant concentration time dependence of these chemicals in the groundwater at a well 100 m downgradient from the disposal facility.

Table 4-5. Concentrations of Selected Chemicals in Well 100 m Downgradient from the Disposal Facility at 1,000 and 10,000 Years after Facility Closure.

Chemical	Performance Goal (mg/L)[a]	Estimated Concentration (mg/L) @ 1,000 y[b]	Estimated Maximum Concentration (mg/L) within First 10,000 y[c]
Chromium	0.1	8.7×10^{-13}	6.1×10^{-3}
Nitrate	10	2.9×10^{-11}	2.0×10^{-1}
Uranium	0.030	0	1.3×10^{-4}

[a] Chemical Performance Goals are from Mann (2002a) and 40 CFR 141
[b] Peak estimated concentration within 1,000 years after facility closure
[c] Peak estimated concentration within 10,000 years after facility closure

Figure 4-8. Time Dependence of Selected Chemical Concentrations in the Groundwater at a Well 100 m Downgradient from the Disposal Facility.



The estimated risks from these chemicals are summarized in Table 4-6. The hazard index and cumulative cancer risk are estimated for the all pathway farmer scenario 100 m downgradient from the disposal site where all water is derived from the well. The estimated impacts provided in Table 4-6 at 1,000 and for the maximum exposure within the first 10,000 years after facility closure for the following chemicals: chromium, nitrate, and uranium. As with the estimated impacts for the radionuclides, the peak impact occurs at approximately 2,400 years after facility closure for chromium and nitrate (K_d = 0 mL/g) and at 10,000 years after facility closure for uranium (K_d = 0.6 mL/g). These estimated risks are compared to established performance objectives (Mann 2002a).

Table 4-6. Estimated Risks from Selected Chemicals from Well 100 m Downgradient from the Disposal Facility at 1,000 and 10,000 Years after Facility Closure.

Type of Impact	Maximum Value within Time Period		Performance Objective
	1,000 years	10,000 years	
Hazard Index (Chemicals)[a]	1.8×10^{-11}	0.12	1.0
Lifetime Cancer Risk (Chemicals)[a]	7.9×10^{-17}	5.6×10^{-7}	10^{-5}

[a] Based on chromium, nitrate, and uranium inventory estimates

4.1.2 SENSITIVITY CASES

The results from the sensitivity case calculations are provided. The base case estimated impacts are provided to demonstrate the impact of a facility design feature that limits the recharge rate into the facility that was assumed for the reference case. The estimated impacts associated with the upper bound inventories for the different waste forms are investigated. Also, the impact of the barrier and barrier degradation are explored for a pulse source.

4.1.2.1 Base Case. The base case assumes a recharge rate of 4.2 mm/y starting at facility closure and extending to all future times. The effect of no barrier on the estimated impacts for the beta/photon dose at a well 100 m downgradient from the IDF disposal facility is shown in Figure 4-9.

The peak dose is estimated to occur at approximately 1,950 years after facility closure. The estimated peak dose from the LLW (Category 1) waste is 0.48 mrem/y and the estimated peak dose from the Category 3 LLW/MLLW is 0.15 mrem/y. At 10,000 years after facility closure, the major contributors to the estimated dose are from the Category 3 LLW and MLLW (42%) and ILAW (58%). Also shown in Figure 4-9 is the total beta/photon dose estimated for the reference case. The peak dose is shifted out approximately 450 years relative to the peak for the base case and the magnitude is approximately the same.

Table 4-7 summarizes the estimated groundwater impacts from the base case for a well 100 m downgradient from the disposal facility. When compared to the results for the reference case we find that the estimated impacts at 1,000 years after facility closure are significantly larger. This is due to the higher recharge for the first 500 years of the base case. The peak impacts for the drinking water dose and the all-pathway scenarios within 10,000 years after facility closure are approximately the same as the reference case.

Figure 4-9. Beta/Photon Drinking Water Dose at Well 100 m Downgradient from the IDF Facility for the Base Case.

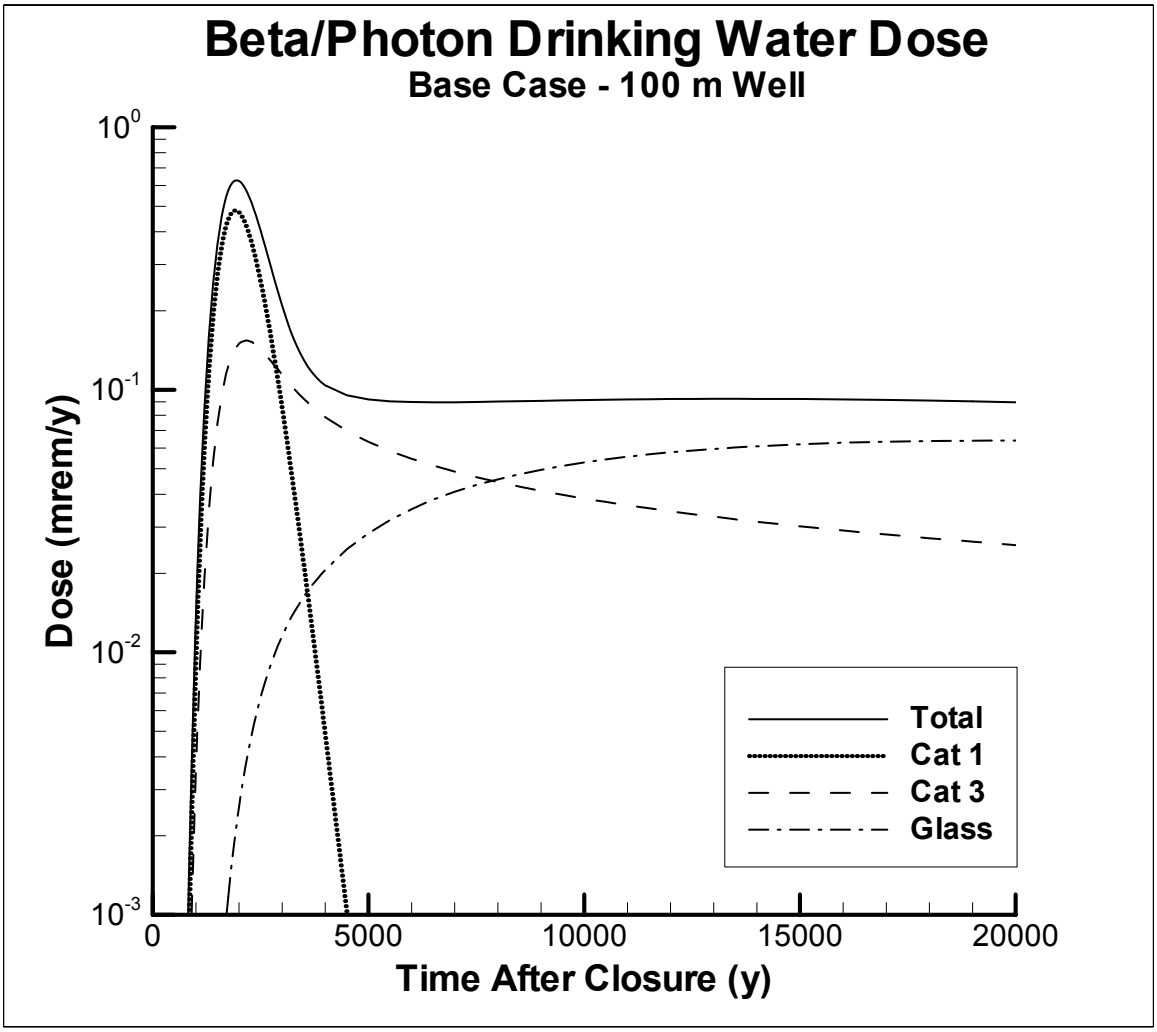


Table 4-7. Estimated Groundwater Impacts from the Base Case for a well 100 m Downgradient from the Disposal Facility.

Type of Impact	Maximum Value within Time Period		Performance Objective
	1,000 years	10,000 years	
Dose (mrem in a year) from beta- and photon-emitting radionuclides in drinking water (Farmer Scenario)	$1.4x10^{-2}$	0.63	4.0
Concentration (pCi/L) of alpha-emitting radionuclides	$5.6x10^{-18}$	0.24	15.0[b]
Farmer Scenario - All-pathway dose (mrem in a year)	$3.6x10^{-2}$	1.62	25.0
Industrial Scenario - All-pathway dose (mrem in a year)	$6.4x10^{-3}$	0.29	25.0
Residential Scenario - All-pathway dose (mrem in a year)	$2.1x10^{-2}$	0.98	25.0
Radium Concentration (pCi/L)	0[a]	0[a]	3.0
Uranium Concentration (pCi/L)	0[a]	0.12	---

[a] The estimated impact is less than $1x10^{-20}$ pCi/L
[b] The performance objective excludes the uranium contribution to the concentration

4.1.2.2 Upper Bound Inventory. Upper bound inventories have been estimated for the ILAW, LLW and MLLW (Tables 3-1 and 3-2). These inventories would be contained in a larger trench (see Table 3-7). The estimated impacts to the all-pathway dose for the farmer scenario is summarized in Table 4-8.

Table 4-8. Estimated Groundwater Impacts for the Upper Bound Inventory for a Well 100 m Downgradient from the Disposal Facility.

Type of Impact	Maximum Value within Time Period		Performance Objective
	1,000 years	10,000 years	
Dose (mrem in a year) from beta- and photon-emitting radionuclides in drinking water (Farmer Scenario)	$1.4e10^{-2}$	1.3	4.0
Concentration (pCi/L) of alpha-emitting radionuclides	$2.1x10^{-25}$	8.7	15.0[b]
Farmer Scenario - All-pathway dose (mrem in a year)	$5.0x10^{-10}$	4.4	25.0
Industrial Scenario - All-pathway dose (mrem in a year)	$6.3x10^{-11}$	0.61	25.0
Residential Scenario - All-pathway dose (mrem in a year)	$2.2x10^{-10}$	2.1	25.0
Radium Concentration (pCi/L)	0[a]	0[a]	3.0
Uranium Concentration (pCi/L)	0[a]	$9.3x10^{-2}$	---

[a] The estimated impact is less than $1x10^{-20}$ pCi/L
[b] The performance objective excludes the uranium contribution to the concentration

4.1.2.3 Pulse/Constant Release. Figure 4-10 shows the contaminant release rates for a 1-year and 1,000 year source pulse for the base case recharge. As the source term length changes from 1 year to 1,000 years the peak flux to the groundwater decreases from 8.7×10^{-4} to 7.2×10^{-4} Ci/y/Ci (~15%) and the time to peak release rate changes from ~1,750 to 2,300 years after facility closure.

Figure 4-10. Pulse Source Term Flux Rates to the Aquifer for the Base Case.

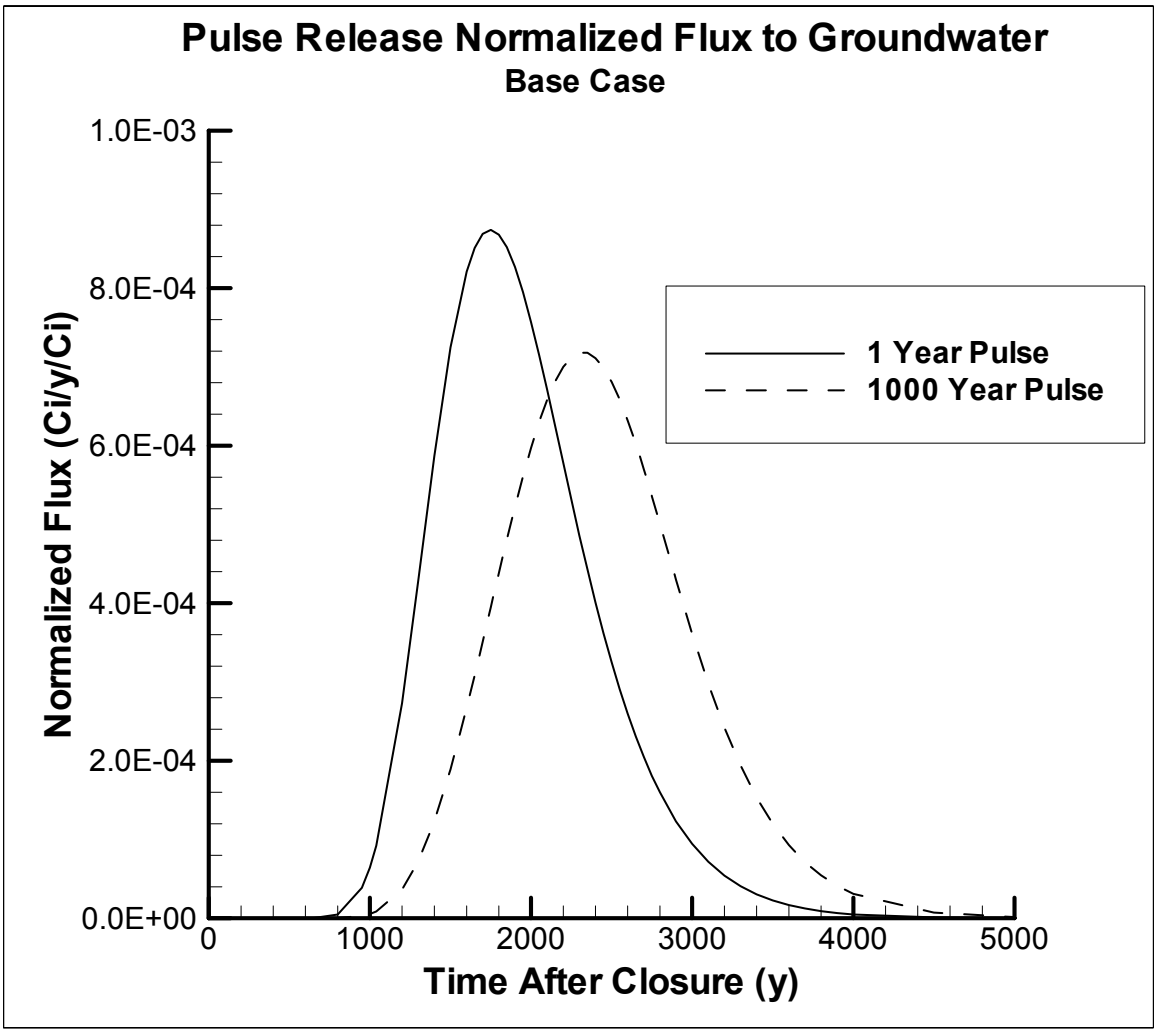


Figure 4-11 shows the impact of the reference recharge rate on the calculated contaminant flux rates to the aquifer for a 1-year, 1000-year and 10,000-year source pulse. Because of the lower recharge for the first 500 years, the contaminant flux to the groundwater decreases from 8.8×10^{-4} to 8.2×10^{-4} Ci/y/Ci (~6.5%) and the time to peak release rate changes from ~2,250 to 2,400 years after facility closure. For a 10,000-year pulse the peak flux to the aquifer is 1.1×10^{-5} Ci/y/Ci.

Figure 4-12 shows the impact of changing the barrier degradation. The reference case assumes the barrier degrades instantaneously at 500 years after facility closure. Figure 4-12 shows the impact if the recharge increases in a step-wise manner from 0.1 to 4.2 mm/y over a 500-year and a 1,500-year period. All three cases are for a 1-year pulse.

Figure 4-11. Pulse Source Term Flux Rates to the Aquifer for the Reference Case.

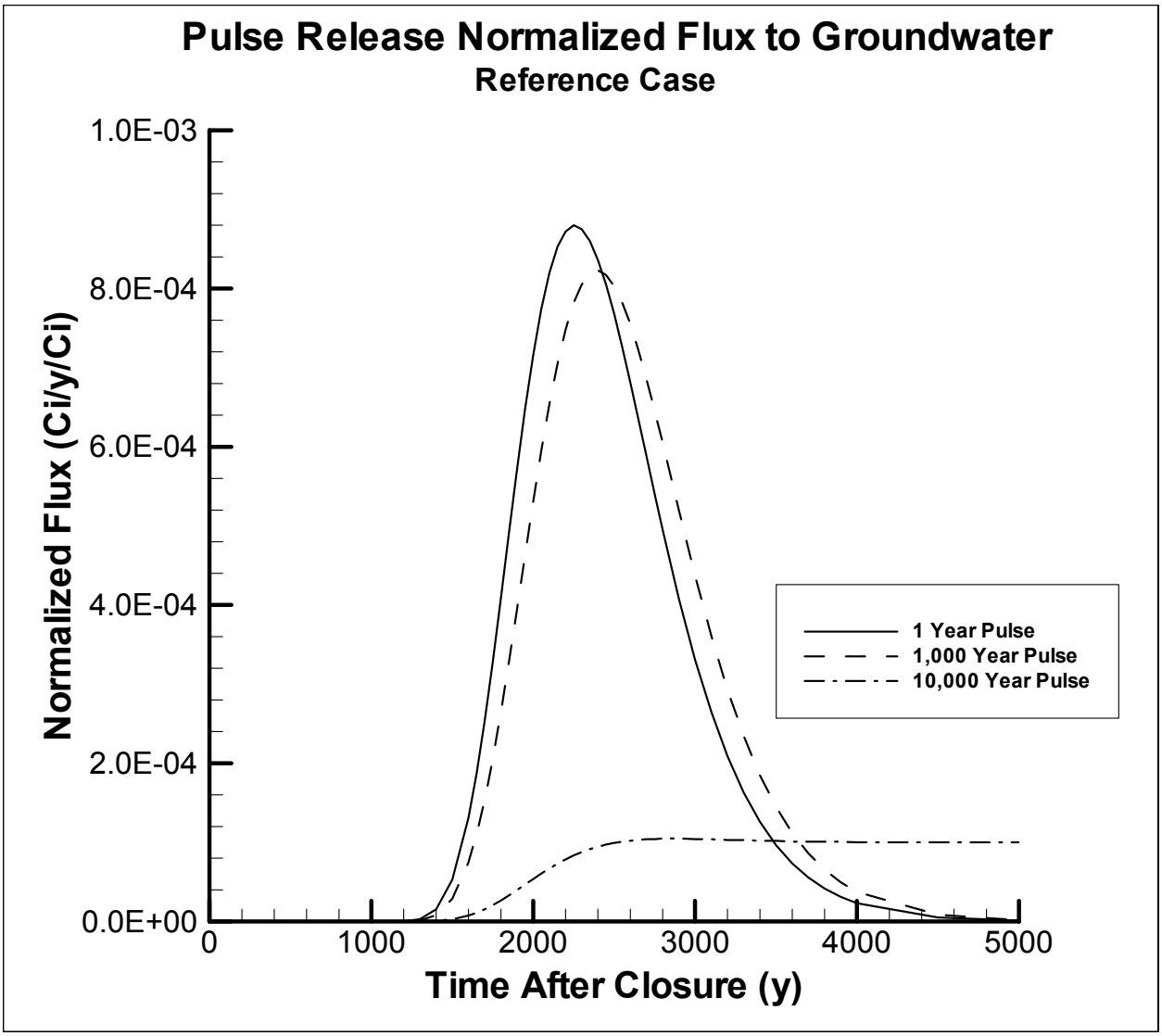
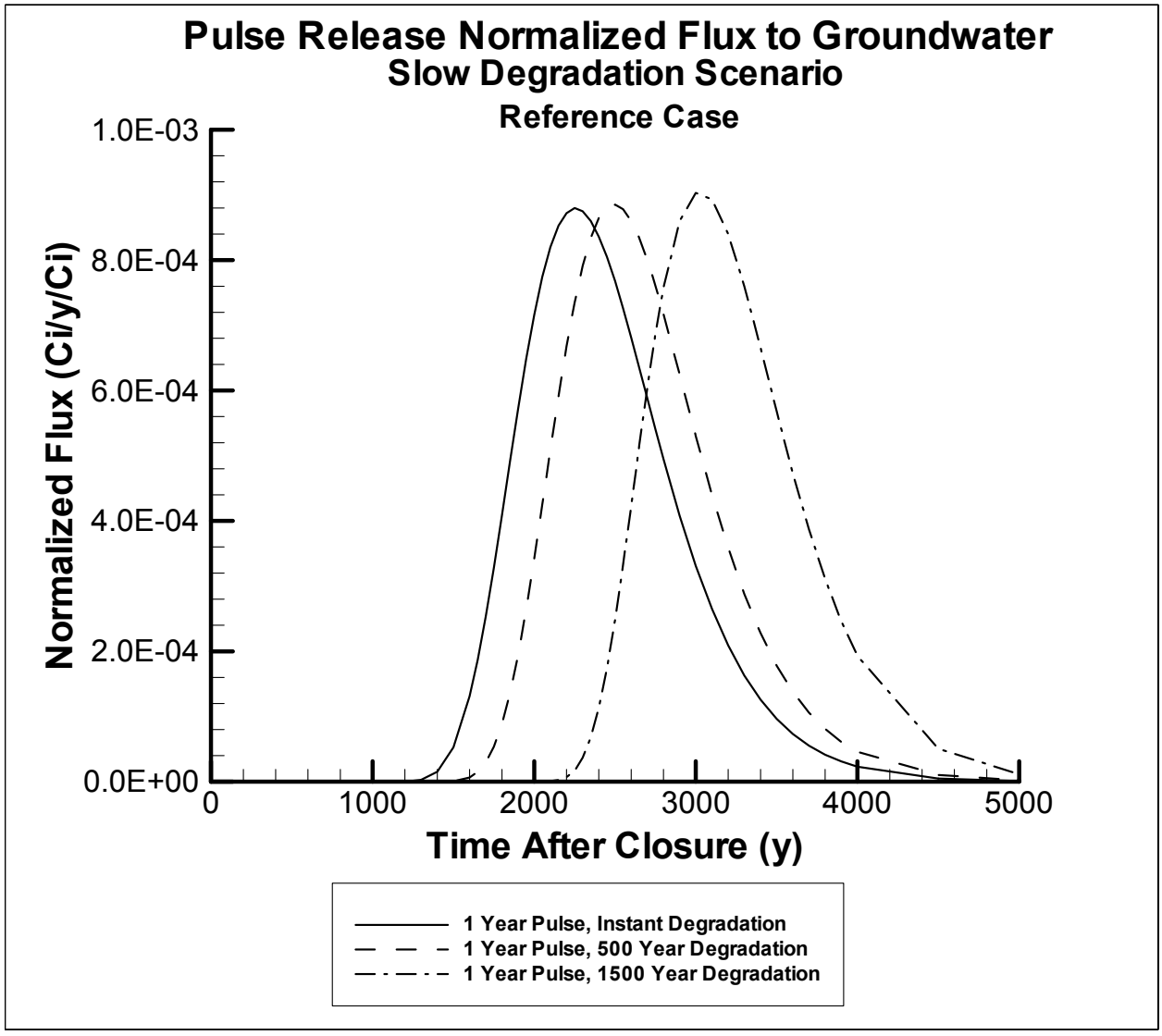


Figure 4-12. Pulse Source Term Flux Rates to the Aquifer for the Reference Case and Different Barrier Degradation Times.



4.2 SURFACE WATER SCENARIO RESULTS

Table 4-9 compares the estimated impacts to the performance objectives for protecting the surface water resources. Estimated impacts are at 1,000 and 10,000 years after facility closure. The estimated impacts are for a well intersecting the groundwater just before the groundwater mixes with the Columbia River. The estimated impacts for the reference case are lower than the performance objectives. Because of the large flow of the Columbia River, mixing will occur in the river and the predicted impacts actually would be far lower than the estimates provided in Table 4-9.

Table 4-9. Estimated Surface Water Impacts from the Reference Case at a Well Just Before Entering the Columbia River.

Type of Impact	Maximum Value within Time Period		Performance Objective
	1,000 years	10,000 years	
Dose (mrem in a year) from beta- and photon-emitting radionuclides in drinking water (Farmer Scenario)	2.3×10^{-12}	3.4×10^{-2}	1.0
Concentration (pCi/L) of alpha-emitting radionuclides	0[a]	9.2×10^{-3}	15.0[b]
Radium Concentration (pCi/L)	0[a]	0[a]	0.3
Uranium Concentration (pCi/L)	0[a]	4.5×10^{-3}	---

[a] The estimated impact is less than 1×10^{-20} pCi/L

[b] The performance objective excludes the uranium contribution to the concentration

4.3 EFFECTS OF RELEASES TO AIR

Earlier PA analyses (Wood et al. 1995a and 1996, Mann et al. 2001) have shown that the effects of contaminant releases to the air are negligible for both ILAW and solid waste (LLW/MLLW) buried in trenches with more than 5 meters of cover above the waste. For this risk assessment we will use the approach developed for these PAs to bound the estimated releases to the air from the waste forms proposed for disposal in the IDF. A superposition approach will be used to estimate the total airborne release and estimated impacts.

4.3.1 Calculational Approach

The principal mechanism by which nuclides migrate from the waste to the ground surface is gaseous diffusion. The analyses in the previous PAs have shown that other mechanisms such as capillary action, upward moisture diffusion, atmospheric pressure and temperature variations, wind, and rainfall have negligible secondary effects on the release of contaminants to the air. The diffusion of radioactive gases such as tritium (as water vapor), ^{14}C (as carbon dioxide), and ^{222}Rn (an inert gas) can be represented using Fick's law of diffusion with a loss term for radioactive decay (Jury 1991). The amount available for diffusion, i.e., the source concentration, is changing with time due to the release mechanism for the contaminants from the waste form and radioactive decay. Two cases (one for tritium and ^{14}C and the other for ^{222}Rn) must be considered because a decay chain that includes ^{238}U, ^{234}U, ^{230}Th, and ^{226}Ra is producing the ^{222}Rn, whereas tritium and ^{14}C are the original source material.

For ^{3}H and ^{14}C the source concentration increases with time after closure due to the release rate of contaminants from the waste form and decreases with time due to radioactive decay. If we first assume that all of the ^{3}H and ^{14}C released during the year diffuses away from the trench, then the source concentration for these two radionuclides is shown below. This assumption will be tested by comparing the predicted total airborne emission rate with the estimated total release rate from the waste form. The two release rates should be comparable in a

system near steady state. The total amount entering the air each year after diffusion through the cover should not be greater than the total amount that is released from the waste form each year.

$$C_0(t) = f\, g\, C_G(0)\, e^{-\lambda t} \tag{4.1}$$

where

$C_0(t)$ = source concentration in the waste trench, i.e., the average concentration of gas that is free to diffuse through the cover soil, Ci/m^3

$C_G(0)$ = average gas concentration in the waste trench at closure (t=0), Ci/m^3. This is computed as the total inventory in the waste trench divided by the trench volume.

f = bounding annual waste release rate fraction. It is the fraction of the total that is released from the waste form.

g = fraction of released contaminant in the gaseous form.

t = elapsed time since closure, y

λ = radioactive decay constant for the nuclide under consideration, per y. It is computed as the natural logarithm of 2 divided by the half-life in years.

The average gas concentration in the waste trench at closure is estimated in the following manner. From the SWBG PA for the 200W area (Wood et al. 1995a) the ^{3}H gas is assumed to be in the form of water vapor in equilibrium with the water within the system. If the released ^{3}H concentration is 1 Ci/m^3, then the water vapor concentration in the waste trench is g = 9.2x10^{-8} Ci/m^3(based on an assumed soil temperature of 10^{o}C, a soil moisture content of 7%, and a total porosity of 17% [corresponding vapor pressure of 0.012 bars]). Finally, the source concentration of ^{222}Rn (g=1) is based on the amount of ^{226}Ra that has been released from the waste form. The ^{226}Ra produces ^{222}Rn by radioactive decay. The amount of ^{226}Ra slowly increases for two reasons. First, an increasing fraction of the inventory is released from the waste form as time increases. Second, ^{226}Ra is being produced by the radioactive decay of ^{238}U and ^{234}U. If we neglect the transport of the uranium from the facility, the peak ^{222}Rn flux occurs after all the contaminants are released from the waste form and the ^{226}Ra has reached radioactive equilibrium with the ^{238}U. This equilibrium occurs after times greater than 1x10^6 years after closure.

The bounding annual waste release fraction will depend on the waste form. For ILAW we have assumed the peak calculated release rate from the waste form (Bacon and McGrail 2002) of 9.3x10^{-7} Ci/y/Ci. For Category 1 LLW we have assumed the annual waste release fraction to be 1. For Category 3 LLW and MLLW we have assumed the waste form is grouted and the bounding annual waste release fraction is given by the initial diffusion rate.

For calculating the diffusion rate in the presence of radioactive decay, the source concentration is assumed to be constant because it changes very slowly with time. This assumption tends to exaggerate the diffusion flux at the surface. The steady-state diffusion equation is shown below. It also assumes the diffusion characteristics of the waste cover are uniform with depth.

$$D\frac{\partial^2 C}{\partial z^2} = \lambda C \tag{4.2}$$

The solution to the above equation has an exponential dependence in elevation as shown below. The boundary conditions that the soil concentration is C_0 at the waste (z=0) and zero at the surface ($z=z_0$) have been included. The solution is only valid from z=0 to $z=z_0$.

$$C = C_0 \left[\frac{e^{-z\sqrt{\lambda/D}} - e^{-(2z_0 - z)\sqrt{\lambda/D}}}{1 - e^{-2z_0\sqrt{\lambda/D}}} \right] \tag{4.3}$$

$$J(z) = -D\frac{\partial C}{\partial z} \quad \text{thus} \quad J(z_0) = \frac{2\sqrt{\lambda D}\, C_0\, e^{-z_0\sqrt{\lambda/D}}}{1 - e^{-2z_0\sqrt{\lambda/D}}} \tag{4.4}$$

where

- C = gas concentration at elevation z in the soil, Ci/m^3. At the bottom of the soil cover, the soil concentration matches the gas concentration in the waste, i.e., $C=C_0$. At the top of the cover the gas concentration is zero.
- D = diffusion coefficient for low atomic number gases moving through soil, 0.01 cm^2/s = 31.56 m^2/y
- J = upward diffusion flux, Ci/m^2 per y
- z = vertical position in the soil, m. The bottom of the soil column is z=0, while the ground surface is $z=z_o$=5 m.
- λ = radioactive decay constant for the nuclide, per y.

Previous calculations did not use the boundary condition that the surface concentration is zero. In effect, an infinite medium was assumed. This underestimates the surface diffusion flux (J) by at least a factor of two. Longer half-life nuclides show a larger difference.

The performance objectives for ^3H and ^{14}C are that the dose downwind be less than 10 mrem/y. The air pathway dose downwind was estimated using the formula shown below.

$$\text{Air Pathway Dose} = J\ A\ T\ DF \tag{4.5}$$

where

- J = flux of the radionuclide from the soil surface, Ci/m^2 per y. Values are given in Table 4-1.
- A = cross-sectional area of the disposal facility trench occupied by the waste form when viewed from above
- T = duration of the release, 1 y
- DF = air pathway dose factor from CAP88-PC for an annual release, mrem

The air transport factor and air pathway dose factors developed in Section 3.2.1 of *Exposure Scenarios and Unit Dose Factors for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment* (Rittmann 1999). The air transport factor is a bounding value that applies to annual emission near the border of a large area source. The dose factors were obtained using the CAP88-PC software from EPA (Parks 1992).

4.3.2 Source Terms

This section defines the diffusion flux to the air for ^{3}H, ^{14}C, and ^{222}Rn for each of the different waste forms. Also provided are the estimated maximum air pathway doses from ^{3}H and ^{14}C for the different waste forms.

4.3.2.1 ILAW. The upper bound ILAW inventory for ^{238}U in the disposal facility is 328 Ci. This estimate is based on the Best Basis Inventory (BBI) estimate reported in Wootan (1999). After one million years, the glass matrix has released the uranium and the ^{230}Th, ^{226}Ra, and ^{222}Rn are all in radioactive equilibrium with the ^{238}U. Thus the ^{222}Rn inventory in the trench due to the ILAW is 328 Ci, and the average concentration in the waste trench is 6.02×10^{-4} Ci/m^3 (assuming the upper bound ILAW waste package loading (81,000 packages) is maximized into one portion of the IDF trench, see Section 3.3.6, Table 3-8). It has been assumed that no migration of the uranium out of the trench has taken place during this time.

The calculation of the diffusion flux for ^{222}Rn is carried out using the formula given in equation 4.4. The diffusion flux at the ground surface is 3.94×10^{-5} Ci/m^2 per year, or 1.25 pCi/m^2 per second. This contribution to the^{222}Rn flux from the IDF is small compared to the performance objective of 20 pCi/m^2 per second.

The air pathway dose calculation is summarized in Table 4-10. The bounding waste inventory at closure is taken from the BBI as reported in Wootan (1999) and corrected for decay to the year 2030. The total dose from these air emissions (0.003 mrem/y) is well below the performance objective of 10 mrem/y. Note that the best estimate ILAW inventory in the waste form does not include either ^{3}H or ^{14}C.

Table 4-10. Calculation Summary for ^{3}H and ^{14}C for ILAW.

	^{3}H	^{14}C
Nuclide Half Life, y	12.33	5,730
Decay Constant, λ, y^{-1}	0.05622	1.210E-04
Bounding Waste Inventory at Closure, A_G (from Table 3-1), Ci	1,322	4,353
Average Trench Concentration at Closure, $C_G(0)$, Ci/m^3	2.43E-03	7.99E-03
Total Waste Release Rate at Closure, f *g* A_G, Ci/y	1.13E-10	4.05E-03
Maximum Source Concentration, $C_0(0)$, Ci/m^3	2.08E-16	7.45E-09
Diffusion Flux into the Air, $J(z_0)$, Ci/m^2 per y	1.30E-15	4.69E-08
Activity Airborne Annually, Ci/y	5.98E-11	2.16E-03

	^{3}H	^{14}C
CAP88-PC Dose Factor, mrem/Ci	0.0237	1.32
Air Pathway Dose, mrem/y	1.42E-12	2.84E-03

Numbers for the radionuclide half-life are from ENDF/B-VI.

Decay constants are the natural logarithm of 2 divided by the half-life (1 y = 365.25 d).

The bounding inventories for ^{3}H and ^{14}C have been decayed 52 years corresponding to the difference between the inventory date (1994) and plant closure (2046).

The average concentration initially in the trenches is calculated as the bounding waste inventory at closure divided by the trench volume required to hold the ILAW inventory, 544,725 m^3.

The total glass release rate is the initial inventory at closure times the bounding annual dissolution fraction, $f = 9.3x10^{-7}$.

The fraction of released contaminant in the gaseous form (g=9.2E-8 for ^{3}H; g=1 for ^{14}C)

The maximum source concentration is calculated from the trench concentration at closure, the bounding annual dissolution fraction, the fraction of released contaminant in gaseous form, and the decay time since closure.

The diffusion flux into the air is calculated from the diffusion model described in the text.

The annual airborne emission from the disposal facility is calculated as the product of the bounding diffusion flux, the facility footprint area corresponding to the trench volume required to hold the ILAW inventory (45,257 m^2), and the release duration (1 y).

Unit release dose factors are from *Exposure Scenarios and Unit Dose Factors for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment* (Rittmann 1999).

The diffusion flux at the surface is shown in Table 4-10. The small values for diffusion flux are largely due to the slow release rate from the glass. The values shown are bounding numbers for the following reasons.

- The estimated glass dissolution rate is $9.3x10^{-7}$ per year after 10,000 years. It is the peak glass dissolution rate for the base analysis case (see Table 4-1 in Mann et al. 2001).
- Only ^{222}Rn is an inert gas that will faithfully follow the diffusion model. The ^{3}H and ^{14}C will undergo chemical reactions and be part of compounds that are likely to exist in the soil or the glass corrosion products. For ^{3}H these compounds include hydroxides. For ^{14}C these compounds include carbonates and carbides. The effect of these chemical reactions is to slow the migration to the surface and the resulting release rate from the ground surface. This effect has not been included in the diffusion calculations to maximize estimated consequences.

4.3.2.1 LLW/MLLW. The LLW and MLLW waste forms can be grouped into three Categories. Category 1 waste is assumed to be immediately available to the environment. Therefore, when considering Equation 4.1, f=1. Category 3 and MLLW are assumed to be grouted. For these two categories the bounding annual release fraction is given by

$$f = \text{integral}\ [\ (A_t/V_t)*\text{sqrt}(D_e/\pi t)]dt = 2*(A_t/V_t)*\text{sqrt}(D_e/\pi) \qquad (4.6)$$

where

A_t = surface area of waste packages (m^2)
V_t = volume of waste packages (m^3)
D_e = diffusion coefficient (m^2/y)
t = time (= 1 y).

Assuming A_t/V_t is equal to a 55-gallon drum, $A_t/V_t = 9.3\ m^{-1}$. We have chosen $D_e = 1x10^{-11}\ cm^2/s = 3.156x10^{-8}\ m^2/y$ based on Kincaid et al. (1995). Therefore, $f = 1.86x10^{-3}$.

The upper bound estimate for ^{14}C in Category 1 LLW includes both organically bound carbon and activated carbon as part of a metal matrix. From the inventory forecasts the upper bound estimate is based predominantly on the disposal of activated steel. For the air release calculations we have assumed the entire inventory is associated with activated steel. If the primary process for ^{14}C release from the steel is corrosion, then the following bounding estimate for ^{14}C release from can be postulated. The volume of metal corroded per year is given by

$$V_c = C * S_m \tag{4.7}$$

Where

V_c = volume of steel corroded (m^3/y)
C = effective corrosion rate for steel (m/y)
S_m = surface area associated with the activated steel disposed in the IDF trench, containing ^{14}C (m^2)

If we let the surface to volume ratio for the steel containing the ^{14}C to be given by

$$\phi = S_m/V_m \tag{4.8}$$

where

V_m = volume of steel containing ^{14}C,

Then,

$$V_c = C * \phi * V_m \tag{4.9}$$

If we assume the ^{14}C is uniformly distributed in the activated steel, then the release rate of ^{14}C from the Category 1 LLW is given by

$$R_c = I_c * V_c/V_m = I_c * C * \phi \tag{4.10}$$

Where

R_c = ^{14}C release rate from the steel (Ci/y)
I_c = inventory of ^{14}C in Category 1 LLW (Ci).

Siciliano (2001) documents the compilation and analysis of all of the Hanford Site references to corrosion effects on the 208-liter drums used for retrievable TRU waste storage, and on other low-carbon steel items buried at the Site. These references provide qualitative and quantitative

historical information that is assembled into one comprehensive knowledge base on low-carbon steel corrosion at the Hanford Site. This total body of data is used to obtain conservative bare-metal corrosion rates for the four onsite storage environments. Provided the storage conditions do not change, these results should continue to be applicable.

The quantitative data analyzed in Siciliano (2001) consists of two different types of measurements: one time, wall thickness measurements of items that have been in service, and in-field controlled studies of specially-prepared steel segments designed to provide corrosion data. These data are analyzed in a complementary and consistent manner to obtain conservative bare-metal corrosion rates, and drum wall thickness distributions. Best value estimates and their corresponding upper bounds obtained for the general corrosion rates are as follows: 0.44 and 0.84 microns/y for aboveground atmospheric shielded storage; 25.8 and 41.1 microns/y for underground soil contact storage; and 11.7 and 19.2 microns/y for underground soil shielded storage at the trench wall. Analysis for drums stored within the interior of the underground soil shielded modules gave the slower rates of 7.9 and 10.6 microns/y. To estimate the release of ^{14}C from activated steel we have assumed the corrosion rate to be 41.1 microns/y (corresponding to the upper bound corrosion rate for the soil contact environment).

To bound the ^{14}C release from the Category 1 LLW a conservative estimate for the surface to volume ratio, ϕ, is needed. The activated steel containing ^{14}C consists predominately of steel components irradiated in reactors. If we assume the surface to volume ratio can be bounded by assuming all the activated steel is sheet metal, then the surface area can be approximated by

$$S_m = 2 * A_m + \Delta \tag{4.11}$$

Where

A_m = the surface area for the sheet
t_m = the sheet thickness
Δ = the surface area associated with the thickness dimension

If we neglect the area contribution due to the thickness dimension, then

$$\phi = (2 * A_m)/(A_m * t_m) = 2/t_m \tag{4.12}$$

If we assume the surface to volume ratio for the activated steel can be bounded by the assumption of sheet geometry with thickness 0.004 m, then the ^{14}C release rate is bounded by

$$R_c = I_c * (41.1 \times 10^{-6} \text{ m/y}) * [2/(0.004 \text{ m})] = 0.0206 \, I_c / y \tag{4.13}$$

The thickness of 0.04 m is based on the observation that the thinnest reactor components included in Category 1 LLW is associated with irradiation vehicle components that have a minimum thickness of 0.004 m (0.15 in.).

The upper bound LLW/MLLW inventory of U in the disposal facility is 760 Ci. After approximately 100,000 years, the LLW and MLLW waste forms have released the uranium and the ^{230}Th, ^{226}Ra, and ^{222}Rn are all in radioactive equilibrium with the ^{238}U. If we assume all the uranium is ^{238}U, then the ^{222}Rn inventory in the trench due to the ILAW is 760 Ci, and the

average concentration in the waste trench is 6.15×10^{-4} Ci/m^3 [assumes V = 123,5000 m^3] (assuming the LLW and MLLW waste package loading is maximized into one portion of the IDF trench, see Table 3-8). It has been assumed that no migration of the uranium out of the trench has taken place during this time.

The calculation of the diffusion flux for ^{222}Rn is carried out using the formula given in equation 4.5. The diffusion flux at the ground surface is 4.02×10^{-5} Ci/m^2 per year, or 1.28 pCi/m^2 per second.

The air pathway dose calculation is summarized in Table 4-11. The bounding LLW/MLLW inventories were taken from Table 3-2.

Table 4-11. Calculation Summary for ^{3}H and ^{14}C in LLW/MLLW.

Relevant Parameters	^{3}H		^{14}C	
	LLW Category 1	LLW Category 3 / MLLW	LLW Category 1	LLW Category 3 / MLLW
Bounding Waste Inventory at Closure, Ci; A_G (from Table 3-2)	1.86E+05	3.97E+01	1.59E+01	1.55E+02
Average Trench Concentration at Closure, Ci/m^3 ; $C_G(0)$	1.50E-02	3.21E-05	1.29E-05	1.29E-04
f	1.00E+00	1.86E-03	1.00E+00	1.86E-03
g	9.20E-08	9.20E-08	2.06E-02	1.00E+00
Total Waste Release Rate at Closure, Ci/y; f g A_G	1.71E-03	6.80E-09	3.30E-01	2.98E-01
Maximum Source Concentration, Ci/m^3 ; $C_0(0)$	1.38E-09	5.50E-15	2.65E-08	2.40E-07
Normalized Diffusion Flux into Air, m^2/y	6.26E+00	6.26E+00	6.31E+00	6.31E+00
Diffusion Flux into the Air, Ci/m^2/y: $J(z_0)$	8.65E-09	3.45E-14	1.68E-07	1.52E-06
Activity Airborne Annually, Ci/y	8.91E-04	3.55E-09	1.73E-02	1.56E-01
Air Pathway Dose, mrem/y	2.11E-05	8.41E-11	2.28E-01	2.06E-01

The bounding inventories for ^{3}H and ^{14}C have been decayed 44 years corresponding to the difference between the inventory date (2002) and plant closure (2046).
The average concentration initially in the trenches is calculated as the bounding waste inventory at closure divided by the trench volume needed to hold the LLW/MLLW inventory, 1,235,000 (=765,000+470,000) m^3.
The bounding release rate is the initial inventory at closure times the bounding annual diffusion rate fraction, 1.86E-3.
The fraction of released contaminant in the gaseous form (g=9.2E-8 for ^{3}H ; g=2.06E-2 for ^{14}C in Cat. 1 waste; g=1 for ^{14}C in Cat 3 waste)
The maximum source concentration is calculated from the trench concentration at closure, the bounding annual dissolution fraction, and the decay time since closure.
The diffusion flux into the air is calculated from the diffusion model described in the text.
The annual airborne emission from the disposal facility is calculated as the product of the bounding diffusion flux, the facility footprint area corresponding to the volume needed to hold the LLW/MLLW inventory (102606 [=63,558+39,048] m^2), and the release duration (1 y).
Unit release dose factors are from *Exposure Scenarios and Unit Dose Factors for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment* (Rittmann 1999).

The estimated air pathway dose is 0.2 mrem/y and is due predominantly to the estimated ^{14}C diffusion from the grouted waste form. This estimated air pathway dose is smaller than the performance objective of 10 mrem/y. This estimate is extremely conservative because it neglects the chemical interactions of the released ^{14}C with the other waste constituents and the soil.

4.3.3 HLW/LAW Melters

The estimated contribution to the air release from the HLW and LAW melters is bounded by the estimated air releases from the grouted LLW/MLLW waste form. The waste remaining in the melters will be grouted and placed into steel overpacks. Table 3-4 provides an estimate of 198 Ci of ^{3}H, 35.2 Ci of ^{14}C and 2.25 Ci of ^{238}U.

Assuming A_t/V_t is equal to the LAW overpack (Table 2-3), $A_t/V_t = 1.215\ m^{-1}$. We have chosen $D_e = 1x10^{-11}\ cm^2/s = 3.156x10^{-8}\ m^2/y$ based on Kincaid et al. (1995). Therefore, $f = 2.43x10^{-4}$.

The upper bound melter inventory of U-238 in the disposal facility is 2.2 Ci. Eventually, the ^{230}Th, ^{226}Ra, and ^{222}Rn are all in radioactive equilibrium with the ^{238}U. The ^{222}Rn inventory in the trench due to the melters is bounded by 2.2 Ci, and the average concentration in the waste trench is $8.76x10^{-5}\ Ci/m^3$ [assumes $V = 25{,}113\ m^3$] (assuming the melter waste package loading is maximized into one portion of the IDF trench). It has been assumed that no migration of the uranium out of the trench has taken place during this time.

The calculation of the diffusion flux for ^{222}Rn is carried out using the formula given in equation 4.4. The diffusion flux at the ground surface is $5.73x10^{-6}\ Ci/m^2$ per year, or $1.8x10^{-1}\ pCi/m^2$ per second.

The air pathway dose calculation is summarized in Table 4-12.

Table 4-12. Calculation Summary for ^{3}H and ^{14}C in Melters.

Relevant Parameters	^{3}H Melter	^{14}C Melter
Bounding Waste Inventory at Closure, Ci; A_G (from Table 3-4)	1.98E+02	3.52E+01
Average Trench Concentration at Closure, Ci/m^3 ; $C_G(0)$	4.24E-04	1.39E-03
f	2.43E-04	2.43E-04
g	9.20E-08	1.00E+00
Total Waste Release Rate at Closure, Ci/y; f g A_G	2.38E-10	8.55E-03
Maximum Source Concentration, Ci/m^3 ; $C_0(0)$	9.47E-15	3.38E-07
Normalized Diffusion Flux into Air, m^2/y	6.26E+00	6.31E+00
Diffusion Flux into the Air, Ci/m^2/y: $J(z_0)$	5.94E-14	2.14E-06
Activity Airborne Annually, Ci/y	1.24E-10	4.45E-03
Air Pathway Dose, mrem/y	2.93E-12	5.88E-03

The bounding inventories for ^{3}H and ^{14}C have been decayed 44 years corresponding to the difference between the inventory date (2002) and plant closure (2046).

The average concentration initially in the trenches is calculated as the bounding waste inventory at closure divided by the trench volume needed to hold the melter inventory, 25,113 (=6400+18,713) m^3.

The bounding release rate is the initial inventory at closure times the bounding annual diffusion rate fraction, 1.86E-3.

The fraction of released contaminant in the gaseous form (g=9.2E-8 for 3H ; g=2.06E-3 for 14C)

The maximum source concentration is calculated from the trench concentration at closure, the bounding annual dissolution fraction, and the decay time since closure.

The diffusion flux into the air is calculated from the diffusion model described in the text.

The annual airborne emission from the disposal facility is calculated as the product of the bounding diffusion flux, the facility footprint area corresponding to the volume needed to hold the LLW/MLLW inventory (2087 [=532+1555] m^2), and the release duration (1 y).

Unit release dose factors are from *Exposure Scenarios and Unit Dose Factors for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment* (Rittmann 1999).

4.3.4 Integrated Estimated Impacts

Table 4-13 summarizes the bounding estimates for the IDF air releases.

Table 4-13. Total Estimate Air Release Impacts.

Estimated Impacts to Facility Air Release	Performance Objective	ILAW	LLW/MLLW	Melters	Total
^{3}H Contribution to Dose (mrem/y)	10 [a]	1.40E-12	2.10E-05	2.90E-12	2.10E-05
^{14}C Contribution to Dose (mrem/y)	10 [a]	2.30E-03	4.34E-01	5.90E-03	4.37E-01
^{222}Rn Concentration ($pCi/m^2/s$)	20	1.25E+00	1.28E+00	1.80E-01	2.71E+00

[a] The performance objective is to have the total estimated dose less than 10 mrem/y from ^{3}H and ^{14}C.

From Table 4-4, we see that the bounding estimates are significantly less than the performance objectives. These estimated impacts are truly bounding for the following reasons:

- Upper bound estimates were used for the inventories for each waste form
- Estimated release rates from the different waste forms were chosen to maximize the release of these contaminants
- No credit was taken for any chemical reactions that would limit the ^{14}C release as a gas (i.e., all the ^{14}C was assumed to be released as a gas)
- The estimated ^{222}Rn flux neglects any movement of the ^{238}U and its daughters from the disposal facility over the time period that the ^{222}Rn inventory builds up (over 100,000 years)
- The estimated ^{222}Rn flux grossly overestimates the available ^{222}Rn inventory produced as a daughter product for times up to 10,000 years after facility closure.

5.0 RESULTS FOR AN INADVERTANT INTRUDER SCENARIO

The intruder scenario analysis is based on the approach and assumptions used in the 2001 ILAW PA. This approach and assumptions have also been used for intrusion into LLW and MLLW. Results are provided assuming the intruder encroached on a section of the IDF where the inventory is either all ILAW or all LLW or MLLW. Special consideration is given to the case where the inventory consists of failed melters from the WTP. The time of compliance is 500 years after closure. However, results will be presented for the period of 100 years to 1,000 years after closure. The performance objective for the driller scenario is 500 mrem (EDE) for a one-time exposure, while the performance objective for the homesteader scenario is 100 mrem (EDE) per year for a continuous exposure. The time of closure is assumed to be 2046. All the analyses presented in this section assume that the surface barrier is in place.

5.1 INADVERTANT INTRUDER SCENARIOS

The pathways described here assume that no memory of the disposal facility remains. An inadvertent intruder (driller scenario) digs or drills into the disposal site and brings some of the waste to the surface, receiving an acute dose. Another intruder (the homesteader scenario) tills the waste into the soil and grows vegetables, receiving a continuous dose while engaged in various activities. The intruder scenario associated with excavating for a basement or building foundation is not considered credible because the top of the waste is 5 meters or more below the surface. Neither basements for homes or foundations for commercial structures are likely to extend this far below the surface. Therefore, this scenario was not evaluated in this risk assessment.

The driller scenario begins with the assumption that some time after disposal operations have ended, a well is drilled through the waste. Drilling at the disposal site is unintentional, and the waste is not recognized as a potential hazard. The waste, along with uncontaminated soil taken from the well, is spread over a work area near the well. The dose to the worker is the sum of the contributions from inhalation of resuspended dust, ingestion of trace amounts of soil, and external exposure at the center of a slab of contaminated soil.

The homesteader scenario considers a family planting a garden using the material taken from the well. Each individual of concern receives dose by direct exposure to the radiation field in the garden, by inhaling resuspended dust, by ingesting trace amounts of soil, and by consuming garden produce. Given that a well is constructed, it is possible, if not probable, that some sort of homesteading will occur.

The scenario of irrigated farming on the disposal site is basically the same as the sensitivity case where the recharge rate is 50 mm/y. This case was not investigated in this risk assessment because this case is not assumed to occur based on the currently planned future uses for the site (see Section 2.1.2).

5.2 ASSUMPTIONS

Selecting values for parameters important in inadvertent intruder scenarios is difficult. Because such intrusion is postulated to be in the future, the nature of the intrusion is ill defined. Moreover, uncertainty abounds about the proper values to be used in a given scenario.

DOE O 435.1 provides no specific guidance on the intruder scenario analysis. For this report the specific exposure scenario for intrusion into ILAW waste is defined in Rittmann (1999) and is generally based on intruder scenarios analyzed in earlier Hanford Site PAs (Wood et al. 1995a, Wood et al. 1996, Mann et al. 1998 and Mann et al. 2001). This specific exposure scenario has also been applied for intrusion into LLW/MLLW.

For the inadvertent intruder scenarios, the most important variables are the amount of waste exhumed, the size of the area over which the waste is spread, and the physical integrity of the waste. Additional parameters, such as exposure time and inhalation rates, also are important but are not typically treated as variables.

For both exposure scenarios, the diameter of the well is assumed to be 0.3 m (1 ft). This value has been used in previous PAs and is larger than the range of well diameters commonly found in local communities (10.2 cm to 25.4 cm [4inches to 10 inches]).

The worker at the well drilling site is assumed to be exposed 8 hours a day for 5 days. The dose to the worker is the sum of the contributions from inhaling resuspended dust (0.12 mg/hour), ingesting trace amounts of soil (100 mg/day), and external exposure at the center of a slab of contaminated soil for 40 hours. The homesteader is assumed to be exposed for 1 year. The soil inhalation rate for the homesteader is 573 mg/year. The incidental ingestion rate is the same as for the driller, 100 mg/day.

The area over which the driller spreads the waste for the driller scenario is assumed to be 100 m^2 (about 1,100 ft^2). This value has been historically used in Hanford Site performance assessments.

The homesteader is assumed to spread the waste over a garden, which is taken to be 200 m^2. In earlier Hanford Site performance assessments, the garden area has been as large as 2,500 m^2 (0.62 acre). The 200 m^2 garden was chosen for the 2001 ILAW PA because the size represents an area large enough to supply a significant portion of a person's vegetable and fruit diet and because the smaller size produces a higher dose, making it the conservative scenario. Household gardens in the vicinity of the Hanford Site range in size from 10 m^2 to 1,000 m^2 (107 ft^2 to 0.25 acre) (Napier et al. 1984). In both scenarios the soil mixing depth is assumed to be 15 cm (5.9 in.). This value has been used in other onsite performance assessments and is the typical rooting depth for garden vegetables.

5.3 INADVERTANT INTRUDER ANALYSIS RESULTS

5.3.1 ILAW Intruder

The assumptions concerning the amount of waste exhumed and the integrity of the ILAW waste form was taken from the 2001 ILAW PA. The amount of ILAW waste material taken from the IDF is assumed to be given by the average waste package loading into the trench. For this risk assessment we have assumed the waste package volume loading into the IDF is 40% of the trench volume. If we neglect the fact that the glass loading into the ILAW package is currently planned to be 85% of the waste package volume then the waste exhumed by the driller can be estimated as: $3.14*(0.305/2)^2$ 13.2 (0.40)= 0.385 m^3.

For this risk assessment we have assumed 99% of the exhumed contaminants remain in the ILAW waste form (i.e., only 1% of the exhumed contaminants are available for ingestion and inhalation).

5.3.1.1 Driller.

The results for the driller scenario are presented in Table 5-1 and displayed in Figure 5-1. The estimated acute exposure dose at 100 years after facility closure is 42.9 mrem. The major contributor to the acute dose is ^{137}Cs, which contributes approximately 94 percent of the dose. ^{90}Sr and ^{126}Sn contribute 2.9 and 2.1 percent, respectively.

The estimated acute exposure dose at 500 years after facility closure (time of compliance) is 1.07 mrem. The major contributor to the acute dose is ^{126}Sn, which contributes approximately 83 percent of the dose. ^{241}Am and ^{237}Np contribute 10 and 4 percent, respectively.

Figure 5-1. Inadvertent Intruder Results for the Driller Scenario - Base Intruder Case into ILAW.

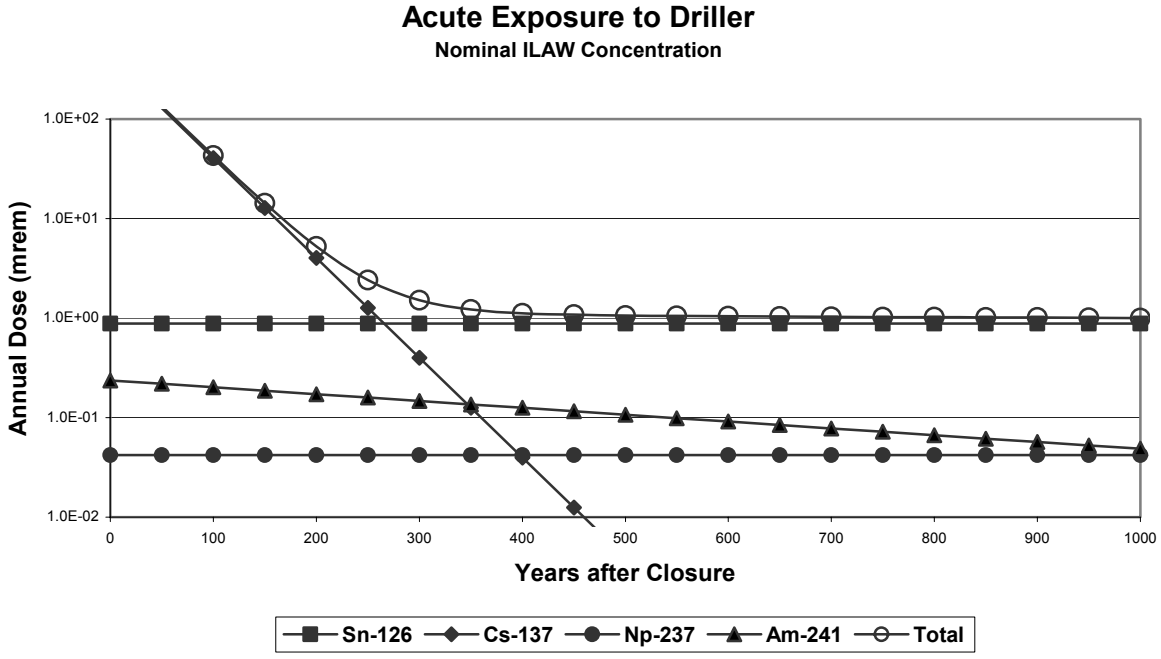


Table 5-1. Driller Intruder Dose (mrem) from ILAW at Various Intrusion Times (years after closure).

Nuclide	100 y	200 y	300 y	400 y	500 y	600 y	800 y	1000 y
Co-60	5.84E-08	1.14E-13	2.21E-19	4.30E-25	8.38E-31	1.63E-36	6.18E-48	2.34E-59
Ni-59	1.91E-08	1.91E-08	1.91E-08	1.91E-08	1.90E-08	1.90E-08	1.90E-08	1.89E-08
Ni-63	1.76E-06	8.83E-07	4.42E-07	2.21E-07	1.11E-07	5.53E-08	1.39E-08	3.47E-09
Se-79	6.67E-07	6.66E-07	6.66E-07	6.66E-07	6.66E-07	6.66E-07	6.66E-07	6.66E-07
Sr-90+D	1.23E+00	1.04E-01	8.91E-03	7.59E-04	6.47E-05	5.51E-06	4.00E-08	2.91E-10
Zr-93	6.61E-05	6.62E-05	6.62E-05	6.62E-05	6.62E-05	6.62E-05	6.62E-05	6.62E-05
Nb-93m	6.30E-08	8.58E-10	1.17E-11	1.59E-13	2.16E-15	2.94E-17	5.44E-21	1.01E-24
Tc-99	1.81E-03	1.81E-03	1.81E-03	1.81E-03	1.81E-03	1.81E-03	1.81E-03	1.81E-03
Ru-106+D	5.37E-46	1.38E-75	3.56E-105	9.15E-135	2.36E-164	6.06E-194	4.02E-253	0.00E+00
Cd-113m	1.83E-06	1.34E-08	9.82E-11	7.20E-13	5.28E-15	3.87E-17	2.08E-21	1.12E-25
Sn-126+D	8.83E-01	8.83E-01	8.83E-01	8.83E-01	8.82E-01	8.82E-01	8.82E-01	8.81E-01
Sb-125	9.86E-16	9.26E-27	8.70E-38	8.17E-49	7.68E-60	7.21E-71	6.36E-93	5.62E-115
I-129	1.44E-04	1.44E-04	1.44E-04	1.44E-04	1.44E-04	1.44E-04	1.44E-04	1.44E-04
Cs-134	1.57E-20	3.94E-35	9.91E-50	2.49E-64	6.26E-79	1.57E-93	9.95E-123	6.29E-152
Cs-137+D	4.05E+01	4.02E+00	3.99E-01	3.96E-02	3.92E-03	3.89E-04	3.83E-06	3.77E-08
Sm-151	1.71E-04	7.92E-05	3.66E-05	1.70E-05	7.85E-06	3.64E-06	7.79E-07	1.67E-07
Eu-152	3.37E-04	1.86E-06	1.03E-08	5.66E-11	3.12E-13	1.72E-15	1.86E-18	1.80E-18
Eu-154	5.79E-04	1.82E-07	5.69E-11	1.79E-14	5.60E-18	1.76E-21	1.73E-28	1.70E-35
Eu-155	4.73E-10	1.75E-16	6.47E-23	2.39E-29	8.84E-36	3.27E-42	4.46E-55	6.09E-68
Ra-226+D	2.49E-04	2.39E-04	2.29E-04	2.19E-04	2.10E-04	2.01E-04	1.84E-04	1.69E-04
Ra-228+D	3.16E-09	1.84E-14	1.07E-19	6.22E-25	3.62E-30	2.10E-35	7.11E-46	2.40E-56
Ac-227+D	4.54E-07	1.88E-08	7.78E-10	3.22E-11	1.33E-12	5.53E-14	9.48E-17	1.63E-19
Th-229+D	2.45E-04	2.43E-04	2.41E-04	2.38E-04	2.36E-04	2.34E-04	2.30E-04	2.25E-04
Th-232	8.51E-03	8.51E-03	8.51E-03	8.51E-03	8.51E-03	8.51E-03	8.51E-03	8.51E-03
Pa-231	3.54E-04	3.56E-04	3.55E-04	3.54E-04	3.54E-04	3.53E-04	3.51E-04	3.50E-04
U-232	3.22E-02	1.19E-02	4.42E-03	1.64E-03	6.07E-04	2.25E-04	3.09E-05	4.23E-06

Table 5-1. Driller Intruder Dose (mrem) from ILAW at Various Intrusion Times (years after closure).

Nuclide	100 y	200 y	300 y	400 y	500 y	600 y	800 y	1000 y
U-233	1.48E-03	2.36E-03	3.24E-03	4.10E-03	4.96E-03	5.81E-03	7.49E-03	9.13E-03
U-234	2.68E-05	4.27E-05	6.58E-05	9.60E-05	1.33E-04	1.76E-04	2.81E-04	4.08E-04
U-235+D	6.56E-04	6.60E-04	6.64E-04	6.68E-04	6.72E-04	6.76E-04	6.84E-04	6.91E-04
U-236	4.19E-07	4.19E-07	4.19E-07	4.19E-07	4.19E-07	4.19E-07	4.19E-07	4.19E-07
U-238+D	2.85E-03	2.85E-03	2.85E-03	2.85E-03	2.85E-03	2.85E-03	2.85E-03	2.85E-03
Np-237+D	4.19E-02	4.19E-02	4.19E-02	4.19E-02	4.19E-02	4.19E-02	4.19E-02	4.20E-02
Pu-238	6.21E-05	2.82E-05	1.28E-05	5.81E-06	2.64E-06	1.21E-06	2.60E-07	6.55E-08
Pu-239	6.70E-03	6.68E-03	6.66E-03	6.64E-03	6.62E-03	6.60E-03	6.56E-03	6.53E-03
Pu-240	1.10E-03	1.09E-03	1.08E-03	1.07E-03	1.06E-03	1.05E-03	1.02E-03	1.00E-03
Pu-241+D	4.58E-03	3.91E-03	3.33E-03	2.84E-03	2.43E-03	2.07E-03	1.51E-03	1.10E-03
Pu-242	9.08E-08	9.08E-08	9.08E-08	9.08E-08	9.08E-08	9.08E-08	9.07E-08	9.07E-08
Am-241	2.02E-01	1.72E-01	1.47E-01	1.25E-01	1.07E-01	9.13E-02	6.66E-02	4.86E-02
Am-243+D	2.94E-04	2.91E-04	2.89E-04	2.86E-04	2.83E-04	2.81E-04	2.75E-04	2.70E-04
Cm-242	1.72E-07	7.80E-08	3.54E-08	1.61E-08	7.32E-09	3.34E-09	7.21E-10	1.81E-10
Cm-243	4.69E-05	4.16E-06	4.05E-07	7.51E-08	4.58E-08	4.30E-08	4.22E-08	4.16E-08
Cm-244	9.53E-07	5.87E-07	5.73E-07	5.66E-07	5.60E-07	5.55E-07	5.43E-07	5.32E-07
Total	4.29E+01	5.26E+00	1.51E+00	1.12E+00	1.07E+00	1.05E+00	1.02E+00	1.01E+00

Note: Nuclides with "+D" added to their name include the contributions from short-lived progeny, which are assumed to be in equilibrium at all times

5.3.1.2 Homesteader.

The results for the homesteader scenario are presented in Table 5-2 and displayed in Figure 5-2. The estimated chronic exposure dose at 100 years after facility closure is 534 mrem. The dose remains above 100 mrem per year out to approximately 200 years. The major contributor to the acute dose at 100 years after facility closure is ^{137}Cs, which contributes approximately 85 percent of the dose. ^{90}Sr and ^{126}Sn contribute 12 and 1.9 percent, respectively. The estimated chronic exposure dose at 500 years after facility closure (time of compliance) is 14.8 mrem. The major contributor to the homesteader dose is ^{126}Sn, which contributes approximately 69 percent of the dose. ^{241}Am and ^{239}Pu contribute 17 and 6 percent, respectively.

Figure 5-2. Inadvertent Intruder Results for the Homesteader Scenario - Base Intruder Case into ILAW.

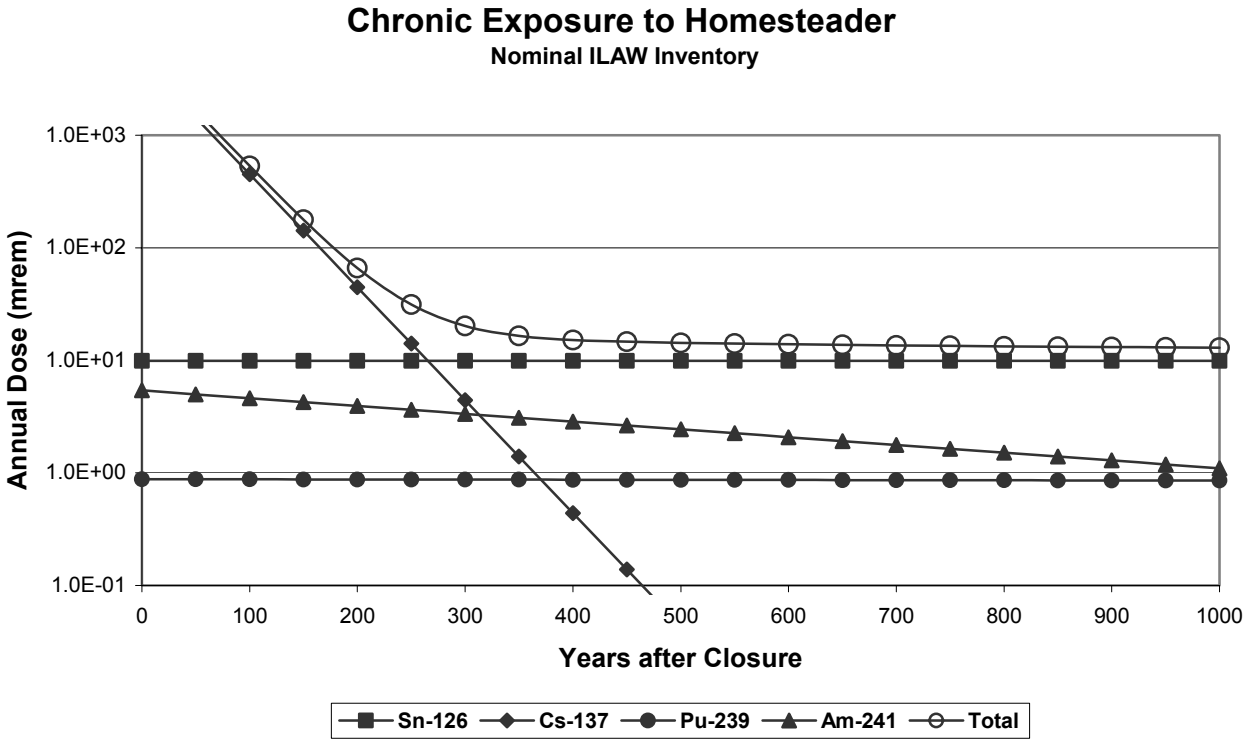


Table 5-2. Homesteader Intruder Dose (mrem/year) from ILAW at Various Intrusion Times (years after closure).

Nuclide	100 y	200 y	300 y	400 y	500 y	600 y	800 y	1000 y
Co-60	6.13E-07	1.19E-12	2.32E-18	4.52E-24	8.81E-30	1.71E-35	6.50E-47	2.46E-58
Ni-59	1.91E-05	1.91E-05	1.91E-05	1.90E-05	1.90E-05	1.90E-05	1.90E-05	1.89E-05
Ni-63	1.76E-03	8.80E-04	4.40E-04	2.20E-04	1.10E-04	5.51E-05	1.38E-05	3.46E-06
Se-79	1.49E-04	1.49E-04	1.49E-04	1.49E-04	1.49E-04	1.49E-04	1.49E-04	1.49E-04
Sr-90+D	6.53E+01	5.56E+00	4.74E-01	4.04E-02	3.45E-03	2.94E-04	2.13E-06	1.55E-08
Zr-93	1.03E-03	1.03E-03	1.03E-03	1.03E-03	1.03E-03	1.03E-03	1.03E-03	1.03E-03
Nb-93m	7.67E-07	1.04E-08	1.42E-10	1.93E-12	2.63E-14	3.58E-16	6.62E-20	1.23E-23
Tc-99	5.71E-01	5.71E-01	5.70E-01	5.70E-01	5.70E-01	5.70E-01	5.69E-01	5.69E-01
Ru-106+D	4.37E-45	1.13E-74	2.90E-104	7.45E-134	1.92E-163	4.94E-193	3.27E-252	0.00E+00
Cd-113m	1.41E-03	1.03E-05	7.55E-08	5.53E-10	4.06E-12	2.97E-14	1.60E-18	8.58E-23
Sn-126+D	9.93E+00	9.92E+00	9.92E+00	9.92E+00	9.92E+00	9.91E+00	9.91E+00	9.90E+00
Sb-125	9.76E-15	9.17E-26	8.61E-37	8.09E-48	7.60E-59	7.14E-70	6.30E-92	5.56E-114
I-129	4.97E-03	4.97E-03	4.97E-03	4.97E-03	4.97E-03	4.97E-03	4.97E-03	4.97E-03
Cs-134	1.50E-19	3.77E-34	9.47E-49	2.38E-63	5.99E-78	1.51E-92	9.51E-122	6.01E-151
Cs-137+D	4.51E+02	4.48E+01	4.44E+00	4.40E-01	4.37E-02	4.33E-03	4.27E-05	4.20E-07
Sm-151	1.22E-02	5.66E-03	2.62E-03	1.21E-03	5.62E-04	2.60E-04	5.57E-05	1.19E-05
Eu-152	3.69E-03	2.04E-05	1.12E-07	6.20E-10	3.42E-12	1.98E-14	9.54E-16	9.54E-16
Eu-154	6.25E-03	1.96E-06	6.15E-10	1.93E-13	6.05E-17	1.90E-20	1.86E-27	1.83E-34
Eu-155	4.95E-09	1.83E-15	6.76E-22	2.50E-28	9.23E-35	3.41E-41	4.66E-54	6.37E-67
Ra-226+D	2.84E-03	2.72E-03	2.61E-03	2.50E-03	2.39E-03	2.29E-03	2.10E-03	1.93E-03
Ra-228+D	3.36E-08	1.96E-13	1.14E-18	6.61E-24	3.84E-29	2.24E-34	7.56E-45	2.55E-55
Ac-227+D	6.28E-06	2.60E-07	1.08E-08	4.46E-10	1.85E-11	7.65E-13	1.31E-15	2.25E-18
Th-229+D	3.02E-03	2.99E-03	2.97E-03	2.94E-03	2.91E-03	2.88E-03	2.83E-03	2.78E-03
Th-232	9.68E-02	9.68E-02	9.68E-02	9.68E-02	9.68E-02	9.68E-02	9.68E-02	9.68E-02
Pa-231	5.17E-03	5.19E-03	5.18E-03	5.17E-03	5.16E-03	5.15E-03	5.13E-03	5.11E-03

Table 5-2. Homesteader Intruder Dose (mrem/year) from ILAW at Various Intrusion Times (years after closure).

Nuclide	100 y	200 y	300 y	400 y	500 y	600 y	800 y	1000 y
U-232	3.64E-01	1.35E-01	4.99E-02	1.85E-02	6.85E-03	2.54E-03	3.48E-04	4.78E-05
U-233	2.65E-02	3.74E-02	4.82E-02	5.89E-02	6.95E-02	7.99E-02	1.01E-01	1.21E-01
U-234	3.09E-03	3.27E-03	3.54E-03	3.89E-03	4.32E-03	4.81E-03	6.02E-03	7.47E-03
U-235+D	7.43E-03	7.49E-03	7.54E-03	7.60E-03	7.66E-03	7.71E-03	7.83E-03	7.94E-03
U-236	9.00E-05	9.00E-05	9.00E-05	9.00E-05	9.00E-05	9.00E-05	9.00E-05	9.00E-05
U-238+D	3.37E-02	3.37E-02	3.37E-02	3.37E-02	3.37E-02	3.37E-02	3.37E-02	3.37E-02
Np-237+D	5.02E-01	5.02E-01	5.02E-01	5.02E-01	5.02E-01	5.02E-01	5.02E-01	5.02E-01
Pu-238	8.33E-03	3.78E-03	1.72E-03	7.80E-04	3.55E-04	1.63E-04	3.55E-05	9.33E-06
Pu-239	8.77E-01	8.75E-01	8.72E-01	8.70E-01	8.67E-01	8.65E-01	8.60E-01	8.55E-01
Pu-240	1.49E-01	1.47E-01	1.46E-01	1.44E-01	1.43E-01	1.41E-01	1.38E-01	1.35E-01
Pu-241+D	1.04E-01	8.91E-02	7.60E-02	6.48E-02	5.52E-02	4.71E-02	3.43E-02	2.50E-02
Pu-242	1.23E-05	1.23E-05	1.23E-05	1.23E-05	1.23E-05	1.23E-05	1.23E-05	1.23E-05
Am-241	4.61E+00	3.93E+00	3.35E+00	2.86E+00	2.43E+00	2.08E+00	1.51E+00	1.10E+00
Am-243+D	3.50E-03	3.46E-03	3.43E-03	3.40E-03	3.37E-03	3.34E-03	3.28E-03	3.22E-03
Cm-242	2.31E-05	1.05E-05	4.76E-06	2.16E-06	9.84E-07	4.50E-07	9.83E-08	2.58E-08
Cm-243	5.54E-04	5.10E-05	6.84E-06	2.95E-06	2.60E-06	2.56E-06	2.54E-06	2.52E-06
Cm-244	1.27E-04	7.90E-05	7.72E-05	7.63E-05	7.55E-05	7.47E-05	7.32E-05	7.16E-05
Total	5.34E+02	6.67E+01	2.06E+01	1.56E+01	1.48E+01	1.44E+01	1.38E+01	1.34E+01

Note: Nuclides with “"+D” added to their name include the contributions from short-lived progeny, which are assumed to be in equilibrium at all times

5.3.2 LLW/MLLW Intruder

The estimated impacts from the inadvertent intruder for intrusion into LLW/MLLW uses the methodology developed for the 2001 ILAW PA (Mann et al. 2001). The LLW/MLLW waste loading into the IDF trench is assumed to be 40% by volume. The amount of waste exhumed by the driller is $(\pi*(.305/2)^2*13.2 * (0.4) = 0.385\ m^3$. The nominal inventory concentration provided in Table 3-2 was used to estimate the exhumed inventory.

The results are based on the nominal inventory estimates provided in Table 3-2 and the average waste volume anticipated for the LLW and MLLW disposed in the IDF trench. For this risk assessment we have assumed all of the exhumed contaminants in the LLW and MLLW are available for ingestion and inhalation.

5.3.2.1 Driller.

The results for the driller scenario are presented in Table 5-3 and displayed in Figure 5-3. The estimated acute exposure at 100 years after facility closure is 1.4 mrem. The major contributor to the acute dose is ^{137}Cs that contributes approximately 72% of the exposure dose. ^{241}Pu contributes approximately 24% of the dose at 100 years after facility closure.

The estimated acute exposure dose at 500 years after facility closure (time of compliance) is 0.23 mrem. The major contributor to the acute dose is ^{241}Pu, which contributes approximately 78% of the dose. ^{226}Ra, ^{239}Pu, and ^{243}Am contribute 5.5, and 9%, respectively, to the acute dose at 500 years after facility closure.

Figure 5-3. Inadvertent Intruder Results for the Driller Scenario - Base Intruder Case into LLW/MLLW.

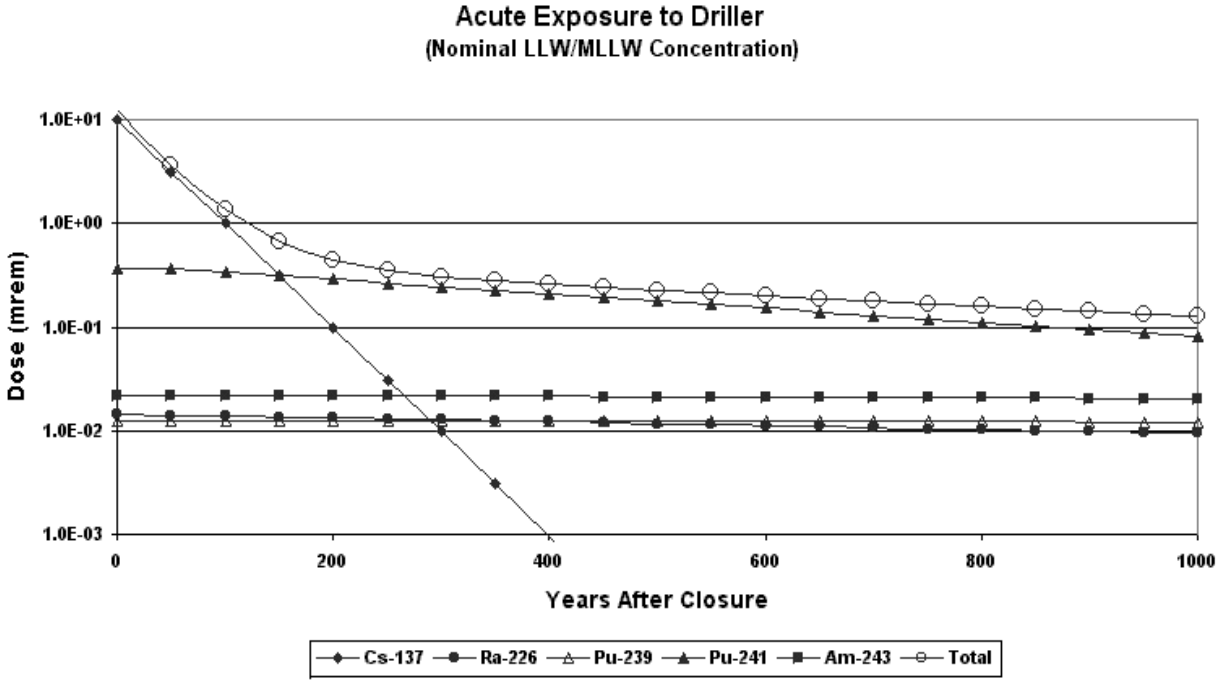


Table 5-3. Driller Intruder Dose (mrem) from LLW/MLLW at Various Intrusion Times (years after closure).

Nuclide	100	200	300	400	500	600	700	800	900	1000
H-3	2.24E-07	8.11E-10	2.94E-12	1.06E-14	3.84E-17	1.39E-19	5.03E-22	1.82E-24	6.59E-27	2.39E-29
C-14	3.39E-07	3.35E-07	3.31E-07	3.27E-07	3.23E-07	3.19E-07	3.15E-07	3.12E-07	3.08E-07	3.04E-07
Co-60	3.96E-06	7.71E-12	1.50E-17	2.92E-23	5.69E-29	1.11E-34	2.16E-40	4.20E-46	8.17E-52	1.59E-57
Se-79	9.19E-07	9.19E-07	9.19E-07	9.19E-07	9.19E-07	9.19E-07	9.19E-07	9.19E-07	9.19E-07	9.19E-07
Sr-90+D	2.36E-03	2.01E-04	1.72E-05	1.46E-06	1.25E-07	1.06E-08	9.05E-10	7.72E-11	6.58E-12	5.61E-13
Tc-99	1.49E-05	1.49E-05	1.49E-05	1.49E-05	1.49E-05	1.49E-05	1.49E-05	1.49E-05	1.49E-05	1.49E-05
I-129	1.44E-04	1.44E-04	1.44E-04	1.44E-04	1.44E-04	1.44E-04	1.44E-04	1.44E-04	1.44E-04	1.44E-04
Cs-137+D	9.92E-01	9.84E-02	9.76E-03	9.68E-04	9.61E-05	9.53E-06	9.46E-07	9.38E-08	9.31E-09	9.23E-10
Ra-226+D	1.39E-02	1.33E-02	1.27E-02	1.22E-02	1.17E-02	1.12E-02	1.07E-02	1.02E-02	9.82E-03	9.40E-03
U-234	3.34E-05	3.40E-05	3.50E-05	3.63E-05	3.78E-05	3.96E-05	4.17E-05	4.40E-05	4.65E-05	4.92E-05
U-235+D	2.97E-05	3.01E-05	3.05E-05	3.09E-05	3.14E-05	3.18E-05	3.22E-05	3.26E-05	3.31E-05	3.35E-05
U-238+D	1.26E-04	1.26E-04	1.26E-04	1.26E-04	1.26E-04	1.26E-04	1.26E-04	1.26E-04	1.26E-04	1.26E-04
Np-237+D	3.72E-03	3.72E-03	3.72E-03	3.72E-03	3.72E-03	3.72E-03	3.72E-03	3.72E-03	3.72E-03	3.72E-03
Pu-239	1.25E-02	1.24E-02	1.24E-02	1.23E-02	1.23E-02	1.23E-02	1.22E-02	1.22E-02	1.22E-02	1.21E-02
Pu-241+D	3.37E-01	2.88E-01	2.45E-01	2.09E-01	1.78E-01	1.52E-01	1.29E-01	1.10E-01	9.39E-02	8.01E-02
Am-243+D	2.20E-02	2.18E-02	2.16E-02	2.14E-02	2.12E-02	2.11E-02	2.09E-02	2.07E-02	2.05E-02	2.04E-02
Cm-243	2.11E-17	7.40E-17	1.61E-16	2.81E-16	4.35E-16	6.21E-16	8.40E-16	1.09E-15	1.37E-15	1.69E-15
Total:	**1.38E+00**	**4.38E-01**	**3.06E-01**	**2.60E-01**	**2.27E-01**	**2.00E-01**	**1.77E-01**	**1.57E-01**	**1.41E-01**	**1.26E-01**

5.3.2.2 Homesteader.

The results for the homesteader scenario intrusion into LLW/MLLW are presented in Table 5-4 and displayed in Figure 5-4. The estimated chronic exposure dose at 100 years after facility closure is 66.3 mrem. The dose remains above 100 mrem per year out to approximately 50 years after facility closure (2046). The major contributor to the acute dose is ^{241}Pu, which contributes approximately 66 percent of the dose. ^{90}Sr and ^{137}Cs contribute 9 and 19 percent, respectively. The estimated chronic exposure dose at 500 years after facility closure (time of compliance) is 26.8 mrem. The major contributor to the homesteader dose is ^{241}Pu, which contributes approximately 85 percent of the dose. ^{239}Pu and ^{243}Am contribute 6 and 4 percent, respectively. While these values satisfy the performance objectives, they are more than an order of magnitude larger than those estimated in the latest annual PA summary for the burial grounds (Wood 2002). This increase is largely due to the more conservative land use assumption used in this analysis (i.e., waste is spread over 200 m^2 rather than 2,500 m^2, thereby decreasing the dilution effect) and a different inventory basis (e.g., in the annual PA summary only previously disposed inventory is considered in the intruder dose calculation).

Figure 5-4. Inadvertent Intruder Results for the Homesteader Scenario - Base Intruder Case into LLW/MLLW.

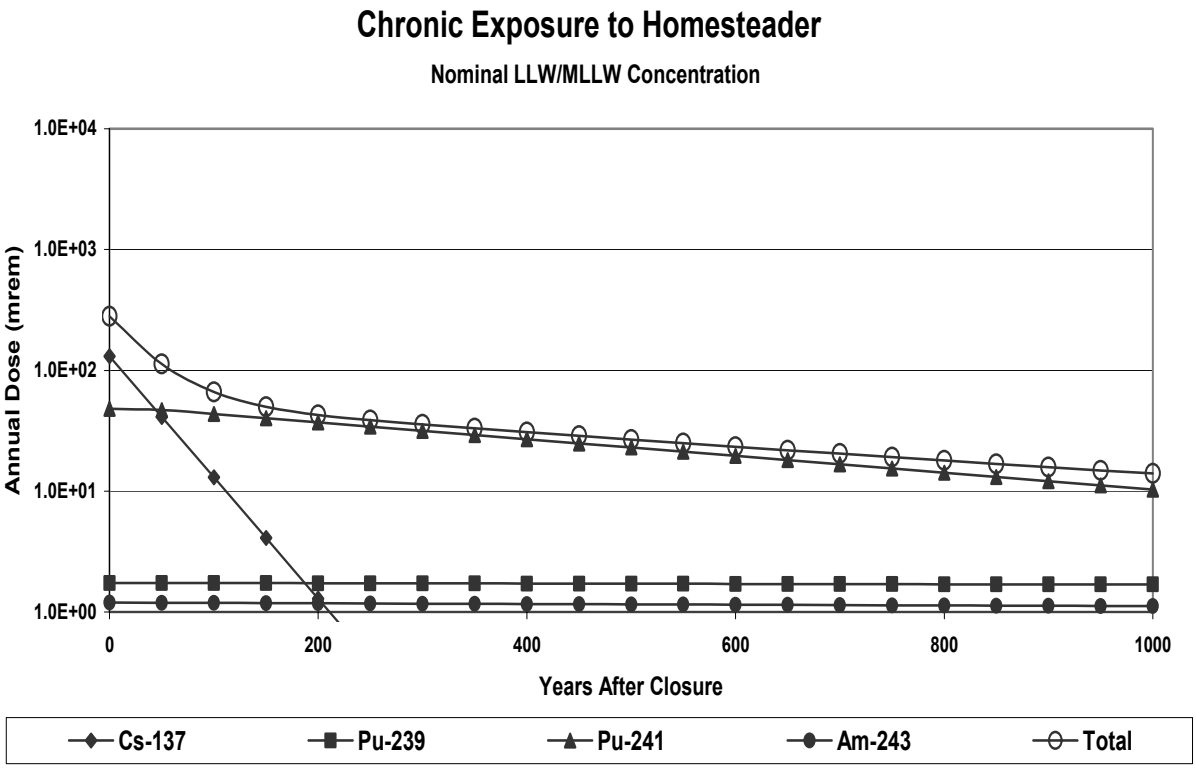


Table 5-4. Homesteader Intruder Dose (mrem/year) from LLW/MLLW at Various Intrusion Times (years after closure).

Nuclide	100 y	200 y	300 y	400 y	500 y	600 y	700 y	800 y	900 y	1000 y
H-3	2.30E-02	8.32E-05	3.01E-07	1.09E-09	3.94E-12	1.43E-14	5.16E-17	1.87E-19	6.76E-22	2.45E-24
C-14	4.96E-03	4.90E-03	4.85E-03	4.79E-03	4.73E-03	4.67E-03	4.62E-03	4.56E-03	4.51E-03	4.45E-03
Co-60	4.19E-05	8.16E-11	1.59E-16	3.09E-22	6.02E-28	1.17E-33	2.28E-39	4.44E-45	8.64E-51	1.68E-56
Se-79	5.76E-04	5.76E-04	5.76E-04	5.76E-04	5.76E-04	5.76E-04	5.76E-04	5.76E-04	5.76E-04	5.76E-04
Sr-90+D	5.85E+00	4.99E-01	4.25E-02	3.62E-03	3.09E-04	2.63E-05	2.24E-06	1.91E-07	1.63E-08	1.39E-09
Tc-99	2.04E-01	2.04E-01	2.04E-01	2.04E-01	2.04E-01	2.04E-01	2.04E-01	2.04E-01	2.04E-01	2.04E-01
I-129	1.10E-01	1.10E-01	1.10E-01	1.10E-01	1.10E-01	1.10E-01	1.10E-01	1.10E-01	1.10E-01	1.10E-01
Cs-137+D	1.30E+01	1.29E+00	1.28E-01	1.27E-02	1.26E-03	1.25E-04	1.24E-05	1.23E-06	1.22E-07	1.21E-08
Ra-226+D	3.54E-01	3.41E-01	3.26E-01	3.12E-01	2.99E-01	2.86E-01	2.74E-01	2.63E-01	2.52E-01	2.41E-01
U-234	1.08E-02	1.08E-02	1.08E-02	1.09E-02	1.09E-02	1.10E-02	1.11E-02	1.11E-02	1.12E-02	1.13E-02
U-235+D	8.47E-04	9.07E-04	9.68E-04	1.03E-03	1.09E-03	1.15E-03	1.21E-03	1.27E-03	1.33E-03	1.39E-03
U-238+D	1.08E-02	1.08E-02	1.08E-02	1.08E-02	1.08E-02	1.08E-02	1.08E-02	1.08E-02	1.08E-02	1.08E-02
Np-237+D	3.07E-01	3.07E-01	3.07E-01	3.07E-01	3.07E-01	3.07E-01	3.07E-01	3.07E-01	3.07E-01	3.07E-01
Pu-238	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-239	1.73E+00	1.73E+00	1.72E+00	1.72E+00	1.71E+00	1.71E+00	1.70E+00	1.70E+00	1.69E+00	1.69E+00
Pu-240	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-241+D	4.35E+01	3.71E+01	3.16E+01	2.69E+01	2.29E+01	1.96E+01	1.67E+01	1.42E+01	1.21E+01	1.03E+01
Pu-242	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Am-241	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Am-243+D	1.19E+00	1.18E+00	1.17E+00	1.16E+00	1.15E+00	1.15E+00	1.14E+00	1.13E+00	1.12E+00	1.11E+00
Total:	**6.63E+01**	**4.28E+01**	**3.56E+01**	**3.08E+01**	**2.68E+01**	**2.33E+01**	**2.04E+01**	**1.79E+01**	**1.58E+01**	**1.40E+01**

5.3.3 Failed WTP Melter Intruder

The potential for an intruder to drill a well such that he includes waste from either a failed ILAW or HLW melter from WTP is considered as a special case. If the driller were to place a well such that he exhumed ILAW waste then the anticipated estimated doses would be bound by the results for ILAW waste (see section 5.3.1).

The current planning for the disposal of the HLW melters from WTP would have these melters placed within 8-inch thick steel containers. Current well drilling techniques would not be capable of penetrating this container.

Moreover, we presume that this container would last over 500 years within the disposal facility due to the low water infiltration into the IDF. Analysis of the total set of underground soil contact (USC) data corrosion rates (Siciliano 2001) gives the best and mean-value bounding values of 25.8 microns/y and 41.1 microns/y for the bare-metal general corrosion rates in general USC environments, i.e., complete soil coverage at random locations and depths. The test shaft data also included a limited amount of coupon bare-metal pitting measurements and analysis of that data gives 112.4 (172.8) microns/y for the best (mean-value bounding) pitting rate values. Assuming the bounding pitting corrosion rate occurs in the IDF for the HLW melter overpack, the estimated maximum pitting depth after 500 years would be 3.4 inches (less than half the available wall thickness). Therefore, for this risk assessment we have assumed that an intruder would not be capable of exhuming HLW glass.

6.0 INTERPRETATION OF RESULTS

6.1 OVERVIEW

This chapter integrates the results presented in Chapters 4 and 5. The numerous results presented in those chapters are reviewed and consolidated to provide a reasoned basis for evaluating the performance of the disposal facility. This interpretation provides a rational basis to conclude that the performance of the disposal facility has been addressed, the analysis is logically interpreted, the results are correct representations of the facility performance, and the results are sufficiently rigorous.

6.2 SUMMARY OF RESULTS

Section 6.3 compares the results to the performance objectives in detail. Section 6.4 compares the results to previous performance assessments. However, based on the results from Chapters 4 and 5, a few general conclusions can be made.

All performance objectives associated with release and migration of radionuclides to the point of compliance are met with a wide margin (ratio of performance objective to predicted value [factor of ~6]). The intruder dose performance objective is met with a smaller margin (factor of ~4). The performance goals associated with release and migration of hazardous chemicals to the point of compliance are met with an even wider margin (factor of ~8) than met by radionuclides.

The groundwater impacts of the three main sources of waste (Category 1 solid waste, Category 3 solid waste, and immobilized low-activity tank waste made into glass) have different temporal shapes. The impacts from Category 1 wastes, which have quick releases, peak early (at ~2,400 years after facility closure for contaminants with K_d = 0 mL/g) and are insignificant after a few thousand years. The impacts from Category 3, which are encased in grout, peak a bit later but in the same general time frame as Category 1 wastes. However, because of the continued release from Category 3 wastes, impacts are still significant at the longest times calculated (20,000 years after facility closure). The impacts from glass are insignificant at the times when the impacts from Category 1 or 3 wastes peak, but the impacts plateau for longer times (greater than 4,000 years after facility closure).

The peak groundwater impacts are due to Category 1 waste with impacts from Category 3 and glass comparable at long times. Because only a relatively few Category 1 packages are expected to drive the results (i.e., those packages with high technetium/ iodine content), the amount of Category 1 waste accepted is quite manageable (e.g., these wastes can be disposed as if they were Category 3 waste). Impacts from melter disposal are not significant relative to the other waste.

The groundwater results for ILAW-glass disposal are about five times higher than those presented in the Hanford Immobilized Low-Activity Waste Performance Assessment: 2001 Version (Mann et al. 2001), but still far below performance objectives. The key drivers are increased Tc-99 inventory due to the removal of the technetium separations process from the Waste Treatment Plant, decreased groundwater dilution due to the placement of the disposal trenches at the south end of the disposal site, and the decrease in contaminant release due to the size of the containers. Additional reductions (such as the effect of using a two-dimensional

model in the near-field [Bacon and McGrail 2002] and better waste form performance [Mann 2002a]) have not been included in this analysis.

The groundwater results for the solid waste disposal are similar to those presented in the latest annual summary (Wood 2002). A straightforward comparison with the burial ground analysis is not possible because several key assumptions affecting contamination estimates are different, leading to both increases and decreases in these estimates. For example, hydrogeologic properties of the unconfined aquifer at this site versus the 200 West Area site create a larger dilution effect and lower the estimated impacts. However, in both cases, performance objectives are easily satisfied.

6.3 COMPARISON OF RESULTS TO PERFORMANCE OBJECTIVES

This section compares the estimated impacts to the performance objectives for each area of protection cited in Section 1.3:

- Protection of the general public
- Protection of the inadvertent intruder
- Protection of groundwater resources
- Protection of surface water resources
- Protection of air resources.

The inadvertent intruder estimated impacts depend on inventory and facility design, and can be mitigated to some extent operationally. The estimated impacts for the other performance objectives depend on inventory, waste form release rate, facility design, and groundwater flow rate (except for air resources).

6.3.1 Protection of General Public

Table 6-1 compares the performance objectives for protecting the general public with the results from the reference case calculations. The estimated all-pathways doses are significantly lower than the performance objectives during the first 10,000 years. At the DOE time of compliance (1,000 years) the estimated impact is insignificant.

The greatest contributors to the peak all-pathways dose are mobile contaminants from the Category 1 wastes, which peak in the few thousand-year time frame (see Figure 6-1). Category 3 wastes show a peak at about the same time. For times exceeding 10,000 years, the contributions from the mobile contaminants from glass, contaminants from Category 3 wastes, and the slightly retarded contaminants from Category 1 wastes are comparable.

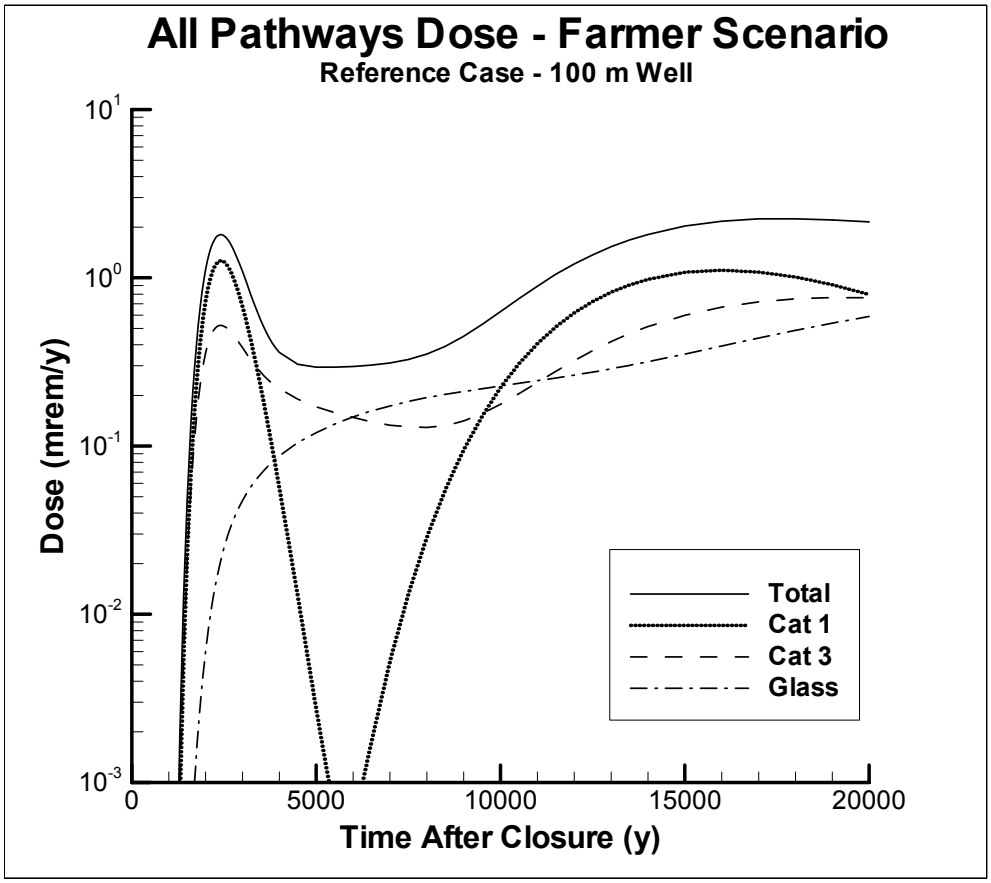
Up to about 5,000 years, the major contributors to the farmer scenario all-path-ways estimated dose are I-129 (~90%) and Tc-99 (~10%). At 10,000 years, Np-237 contributes 44% of the all-pathways dose, Tc-99 contributes 35%, I-129 contributes 17%, and other radionuclides contribute 4%.

Table 6-1. Comparison of Estimated Impacts with Performance Objectives for Protecting the General Public. The DOE time of compliance is 1,000 years.

Performance Measure	Performance Objective	Estimated Impact at 1,000 years	Estimated Impact During First 10,000 years
All-pathways [mrem in a year]	25.0		
Farmer Scenario		1.2×10^{-10}	1.8
Residential Scenario		0.73×10^{-10}	1.1
Industrial Scenario		0.22×10^{-10}	0.32
ILCR (Chemicals)[a]	10^{-5}	7.9×10^{-17}	5.6×10^{-7}
HI (Chemicals)[a]	1.0	1.8×10^{-11}	0.12

[a] Based on chromium, nitrate and uranium inventory estimates
ILCR Incidental lifetime cancer risk
HI Hazard index

Figure 6-1. Time Dependence of the Estimated Farmer Scenario All-Pathways Dose at a Well 100 m Downgradient from the Disposal Facility



6.3.2 Protection of the inadvertent intruder

Table 6-2 compares the estimated impacts to the performance objectives for protecting the inadvertent intruder. The time of compliance starts at 500 years after closure. The acute exposure performance objective is met by a factor of ~500. The maximum acute exposure dose is based on all the exhumed waste being ILAW. Sn-126 is the most important radionuclide. The continuous exposure performance objective is met by a factor of approximately four for the reference case. The maximum homesteader dose is based on all the exhumed waste being LLW/MLLW. Pu-241, Am-243, and Pu-239 are the major contributors. These results are similar in magnitude to those found in the ILAW PA (Mann et al. 2001).

The estimated impacts for the inadvertent intruder can be mitigated through operational controls based on projected container inventories. Such operational controls will be better defined as the project matures.

Table 6-2. Comparison of Estimated Impacts with Performance Objectives for Protecting the Inadvertent Intruder. The time of compliance starts at 500 years.

Performance Measure	Performance Objective	Estimated Impact at 500 years
Acute exposure [mrem]	500.0	1.06
Continuous exposure [mrem in a year]	100.0	26.8

6.3.3 Protection of groundwater resources

Table 6-3 compares the estimated impacts to the performance objectives for protecting the groundwater resources. At the DOE time of compliance (1,000 years) and the point of compliance (at a well 100 m downgradient of the disposal facility), the groundwater impacts are not significant. For the first 10,000 years the estimated impacts are approximately a factor of six less than the performance objectives for beta-photon emitters and a factor of 150 less than the performance objectives for the alpha-emitting radionuclides for the reference case. The concentration of radium is insignificant.

The most important isotopes are those for the all-pathways.

Table 6-3. Comparison of Estimated Impacts with Performance Objectives for Protecting Groundwater. The DOE time of compliance is 1,000 years.

Performance Measure	Performance Objective	Estimated Impact at 1,000 years	Estimated Impact During the First 10,000 years
βγ Emitters [mrem/year]	4.0	4.7×10^{-11}	0.70
Alpha-emitters [pCi/L]	15.0[a]		
All radionuclides		0[b]	0.19
Non-uranium radionuclides		0[b]	0.10
Ra [pCi/L]	5.0	0.0[b]	0.0[b]

[a] The performance objective excludes uranium contribution to the concentration
[b] The estimated impact at 1,000 years after facility closure was less than 1×10^{-20} pCi/L

6.3.4 Protection of Surface Water Resources

Table 6-4 compares the estimated impacts to the performance objectives for protecting the surface water resources. The DOE time of compliance is 1,000 years and the point of compliance is at a well intercepting the groundwater just before it mixes with the Columbia River. The estimated impacts at 1,000 years are insignificant. The impacts during the first 10,000 years are far below performance objectives.

Table 6-4. Comparison of Estimated Impacts with Performance Objectives for Protecting Surface Water Resources (Base Analysis Case). The DOE time of compliance is 1,000 years. The point of compliance is a well intercepting the groundwater before entering the Columbia River.

Performance Measure	Performance Objective	Estimated Impact at 1,000 years	Estimated Impact During the First 10,000 years
All-Pathways [mrem/y]	25.0		
Farmer scenario		6.1×10^{-12}	0.089
Native American		2.54×10^{-11}	0.35
βγ Emitters [mrem/y]	1.0	2.3×10^{-12}	0.034
Alpha Emitters [pCi/L]	15.0[a]	0[b]	0.0092
Ra [pCi/L]	3.0	0.0[b]	0.0[b]

[a] The performance objective excludes uranium contribution to the concentration
[b] The estimated impact at 1,000 years after facility closure was less than 1×10^{-20} pCi/L

6.3.5 Protection of Air Resources

Table 6-5 compares the estimated impacts to the performance objectives for protecting air resources. The DOE time of compliance is 1,000 years and the point of compliance is just above the disposal facility. The estimated impacts are significantly lower than the performance objectives and are based on extremely conservative assumptions.

Table 6-5. Comparison of Estimated Impacts with Performance Objectives for Protecting Air Resources. The DOE time of compliance is 1,000 years. The point of compliance is just above the disposal facility.

Performance Measure	Performance Objective	Estimated Impact at 1,000 years
Radon [pCi m^{-2} $second^{-1}$]	20.0	2.7
Other radionuclides (^{3}H and ^{14}C) [mrem in a y]	10.0	0.44

6.4 COMPARISON OF RESULTS TO PREVIOUS PERFORMANCE ASSESSMENTS

The wastes analyzed in this risk assessment have been analyzed before in the Hanford Immobilized Low-Activity Waste Performance Assessment: 2001 Version (Mann et al. 2001) and Performance Assessment for the Disposal of Low-Level Waste in 200 West Area Burial Grounds (Wood et al. 1995a). This section shows that based on recent decisions, the impacts estimated here for ILAW waste are higher than in the 2001 ILAW PA, but are still significantly lower than performance objectives. For solid waste, the impacts estimated here are higher than estimated in the 200 West Area Solid Waste PA because of higher inventory estimates. However, the results are lower than the most recent annual summary (Wood 2002). A direct comparison of this analysis with the burial ground analyses is not straightforward because of numerous differences in assumptions affecting groundwater contamination estimates. However, because of greater groundwater flow underneath the IDF than under the 200 West Area Solid Waste Burial Grounds, movement of the solid waste disposal site from the 200 West Area to the IDF site will result in lower impacts to the groundwater for those wastes.

Since the analysis was performed for the 2001 ILAW PA (Mann et al. 2001), several important decisions have been made which affect the estimation of impacts:

- Changes in Tc-99 inventory
- Placement of ILAW on disposal site
- Size of ILAW containers.

The first two of these were explicitly analyzed in the 2001 ILAW PA.

The 2001 ILAW PA assumed that the Waste Treatment and Immobilization Plant (WTP) would contain a Tc-99 separations stage that would result in only 20% of the Tc-99 going into ILAW. The Field Manager of ORP has approved the elimination of the Tc-separations process in WTP. This increases the amount of Tc-99 by a factor of 5. Because of the presence of I-129, the beta/gamma drinking water dose increases by a

factor of 3 while the all-pathways dose at 10,000 years increases less because of the presence of not only I-129, but also slightly retarded contaminants such as Np-237. The 2001 ILAW PA assumed that the ILAW trenches would be at the north end of the disposal site. Current designs have the IDF trench located more towards the southern end of the site. This results in less groundwater dilution, resulting in groundwater-pathway impacts increasing by a factor of 2. Finally, the size of containers has increased. This slightly decreases groundwater-pathway impacts (due to the slightly smaller contaminant release rates estimated by Bacon and McGrail [2002]), while slightly increasing the inadvertent intruder impacts (due to increasing the quantity of waste exhumed). However, overall the estimated impacts are still significantly below performance objectives.

6.5 PERFORMANCE SENSITIVITY TO KEY PARAMETER UNCERTAINTIES

The key uncertainties of this analysis are as follows:

- Uncertainties in inventory
- Uncertainties in release rates from Category 3 and ILAW
- Uncertainties in retardation for slightly retarded contaminants from Category 1 waste
- Uncertainties in recharge
- Uncertainties in groundwater flow.

The greatest groundwater pathway impacts are from Category 1 and Category 3 solid waste disposal. The inventory for these wastes is quite uncertain since they depend on future decisions. In particular, the amount of offsite waste to be disposed at Hanford as a result of the Solid Waste EIS Record of Decision is uncertain. Refinement is also expected from WTP secondary waste streams as the WTP contractor finishes design and as operations begin.

For long time periods (i.e., over 5,000 years), the impacts are sensitive to the release rates from Category 3 wastes and from ILAW. The release rate from Category 3 waste was estimated based on a representative diffusion coefficient. The use of an effective diffusion model to represent the release rate of contaminants from grouted LLW and MLLW needs to be investigated further. Work is continuing on ILAW release rates and as shown by the latest ILAW annual summary (Mann 2002b), the ILAW release rates used here are conservative.

Interestingly, the slightly retarded contaminants from Category 1 waste have similar estimated impacts from the mobile contaminants from Category 3 wastes and ILAW. The retardation factor for the slightly retarded contaminants is based on the lowest values thought to be likely in the Hanford environment. More realistic values for retardation, if appropriate, would lower the estimated impacts.

Although not explicitly modeled in this document, the 2001 ILAW PA showed the strong dependence of estimated impacts on the rate at which water leaves the ground

surface and enters the disposal facility (i.e., the recharge rate). Again, conservative values were used in this analysis. Better estimates should lower estimated impacts.

6.6 SUMMARY

All of the estimated impacts easily meet the performance objectives set out in Section 1.3. The estimated all-pathways dose, beta-photon drinking water dose, and concentration of alpha-emitting radionuclides in groundwater for the reference case are more than a factor of 6 lower than the corresponding performance objective at 10,000 years after facility closure (2046). This margin increases by many orders of magnitude if the time of compliance of 1,000 years is used. These estimates are conservatively calculated and hence should provide reasonable expectation that human health and the environment will be protected.

7.0 RECOMMENDATIONS

Based on the analyses in this document as well as the preceding analyses, there is reasonable expectation that the disposal of future radioactive solid waste (LLW and MLLW), immobilized low-activity waste from the WTP, and melters from the WTP will be protective of long-term human health and the environment. The analyses presented here are conservative and more refined analyses (such as a planned performance assessment to satisfy the requirements of the DOE order on radioactive waste management) should provide lower estimates.

Given the set of assumptions used in this analysis, primary results are 1) moving the disposal of future solid waste from the 200 West Area to the IDF leads to lower estimates because of greater groundwater dilution, 2) given recent decisions on ILAW (e.g., the increase of ^{99}Tc going to ILAW), the impacts for ILAW disposal increase but are still significantly below performance objectives and below the estimated impacts from the disposal of solid waste at the same site, and 3) very long-term impacts (~10,000 years) are about equally shared among slightly retarded contaminants from Category 1 waste, contaminants from Category 3 waste, and mobile contaminants from ILAW.

The largest uncertainties in this conservative analysis come from inventory and release rate unknowns. Much of the uncertain in inventory is driven by future management decisions. Because of the importance of Category 1 and to a lesser extent to Category 3 wastes, the inventory projections for these inventories must be investigated further. In particular, how much of the iodine-129 can be treated to become Category 3 wastes should be understood.

The uncertainty for release rates is being addressed by Hanford performance assessment activities. The ILAW PA activity is investigating the release rates of WTP ILAW glasses (Mann 2002b). The solid waste burial ground PA activity is investigating the diffusion rates for grouts currently being used for Category 3 wastes (Wood 2002).

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Dr. Mann is the team leader for the ILAW Performance Assessment activity, which is charged with preparing this document. He was the main author of the 1998 and 2001 versions of the Hanford *Immobilized Low-Activity Tank Waste Performance Assessment*. He has worked for over 25 years in the field of nuclear data and the application of those data to large energy facilities. He has advised the DOE and the International Atomic Energy Agency. He is one of the main authors of this document.

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[1. Page 9 discusses use the central plateau wisely for waste management.]

[2. Page 23 discusses the future use options of the 200 Areas.]

[3. Page 25 discusses the cleanup scenario.]

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ATTACHMENT 1 - TECHNICAL PEER REVIEW CHECKLISTS

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Desk Instruction 2.0, Rev. 1

CHECKLIST FOR TECHNICAL PEER REVIEW

Document Reviewed - Paul Rittmann

Title: Integrated Disposal Facility Risk Assessment
Scope: all except VAM input + output

Author: FM Mann, RJ Puigh, S Finfrock, R Khaleel, M Wood

Date: April 29, 2003

Yes	No*	NA		
[✓]	[ ]	[ ]		Referenced analyses appropriate.
[✓]	[ ]	[ ]		Problem completely defined and all potential configurations considered.
[✓]	[ ]	[ ]		Accident scenarios developed in a clear and logical manner.
[✓]	[ ]	[ ]		Necessary assumptions explicitly stated and supported.
[✓]	[ ]	[✓]		Computer codes and data files documented.
[✓]	[ ]	[ ]		Data used in calculations explicitly stated in document.
[✓]	[ ]	[ ]		Data checked for consistency with original source information as applicable.
[✓]	[ ]	[ ]		Mathematical derivations checked including dimensional consistency of results
[✓]	[ ]	[ ]		Models appropriate and used within range of validity, or use outside range of established validity justified.
[✓]	[ ]	[ ]		Hand calculations checked for errors. Spreadsheet results should be treated exactly the same as hand calculations.
[ ]	[ ]	[✓]		Software input correct and consistent with document reviewed.
[ ]	[ ]	[✓]		Software output consistent with input and with results reported in document reviewed.
[✓]	[ ]	[ ]		Limits/criteria/guidelines applied to analysis results are appropriate and referenced. Limits/criteria/guidelines checked against references.
[✓]	[ ]	[ ]		Safety margins consistent with good engineering practices.
[✓]	[ ]	[ ]		Conclusions consistent with analytical results and applicable limits.
[✓]	[ ]	[ ]		Results and conclusions address all points required in the problem statement.
[✓]	[ ]	[ ]		Format consistent with applicable guides or other standards.
[ ]	[ ]	[✓]	**	Review calculations, comments, and/or notes are attached.
[✓]	[ ]	[ ]		Document approved (for example, the reviewer affirms the technical accuracy of the document).

Paul Rittmann Paul Rittmann — Technical Peer Reviewer (printed name and signature)

5-7-03 — Date

* All "no" responses must be explained below or on an additional sheet.

** Any calculations, comments, or notes generated as part of this review should be signed, dated and attached to this checklist. The material should be labeled and recorded in such a manner as to be understandable to a technically qualified third party.

Desk Instruction 2.0, Rev. 1

CHECKLIST FOR TECHNICAL PEER REVIEW

Document Reviewed - Mel Piepho

Title: Integrated Disposal Facility Risk Assessment

Author: FM Mann, RJ Puigh, S Finfrock, R Khaleel, M Wood

Date: ~~April 29~~ May 7, 2003

Yes	No*	NA		
[✓]	[ ]	[ ]		Referenced analyses appropriate.
[✓]	[ ]	[ ]		Problem completely defined and all potential configurations considered.
[ ]	[ ]	[✓]		Accident scenarios developed in a clear and logical manner.
[✓]	[ ]	[ ]		Necessary assumptions explicitly stated and supported.
[ ]	[ ]	[✓]		Computer codes and data files documented. VAM Input/output files on CD
[✓]	[ ]	[ ]		Data used in calculations explicitly stated in document.
[ ]	[ ]	[✓]		Data checked for consistency with original source information as applicable.
[ ]	[ ]	[✓]		Mathematical derivations checked including dimensional consistency of results
[✓]	[ ]	[ ]		Models appropriate and used within range of validity, or use outside range of established validity justified.
[ ]	[ ]	[✓]		Hand calculations checked for errors. Spreadsheet results should be treated exactly the same as hand calculations.
[✓]	[ ]	[ ]		Software input correct and consistent with document reviewed.
[✓]	[ ]	[ ]		Software output consistent with input and with results reported in document reviewed.
[ ]	[ ]	[✓]		Limits/criteria/guidelines applied to analysis results are appropriate and referenced. Limits/criteria/guidelines checked against references.
[ ]	[ ]	[✓]		Safety margins consistent with good engineering practices.
[ ]	[ ]	[✓]		Conclusions consistent with analytical results and applicable limits.
[ ]	[ ]	[✓]		Results and conclusions address all points required in the problem statement.
[ ]	[ ]	[✓]		Format consistent with applicable guides or other standards.
[✓]	[ ]	[ ]	**	Review calculations, comments, and/or notes are attached. 2 pages
[✓]	[ ]	[ ]		Document approved (for example, the reviewer affirms the technical accuracy of the document).

M. G. Piepho Mel D. Piepho — Technical Peer Reviewer (printed name and signature)

5/7/03 — Date

* All "no" responses must be explained below or on an additional sheet.

** Any calculations, comments, or notes generated as part of this review should be signed, dated and attached to this checklist. The material should be labeled and recorded in such a manner as to be understandable to a technically qualified third party.

Comments on RPP-15834, Revision F and VAM3DF Input Files

By Mel Piepho, May 7, 2003

The documentation inconsistencies have been cleared up for the final revision of RPP-15834, but the details from an earlier draft are still shown below because some of inconsistencies are in the VAM3DF documentation.

In the VAM3DF documentation, the apparent molecular diffusion coefficient, D^0, is equal to the tau x porosity x D^*, where D^* is the molecular diffusion coefficient in pore water and has a reference value of 2.5 x 10^{-5} cm^2/sec in RPP-15834. Furthermore, the VAM3DF documentation suggests that tau, the tortuosity, can be set to the cube root of the porosity as an approximation. The soil with gravel in the model was assigned a value of 0.375 and the sandy soil was assigned a porosity value of 0.138, and the corresponding tortuosity or cube root is 0.721 for the sandy soil and 0.517 for the gravelly soil.

Hence, the input value of D^0 should be 2.13E-2 m2/yr for the sandy soil in card GROUP 11 for VAM3DF, and 5.63E-3 m2/yr for the gravelly soil, which are the actual values appearing in all of the input files.

However, there is an inconsistency in VAM3DF document on page 2-23 and the card GROUP 11 input for D^0. Equation 2.27a on page 2-23 shows the tortuosity factor outside of D^0, whereas card GROUP 11 shows the tortuosity factor as part of the D^0. It appears as if the tortuosity is separated from D^0 in Equation 2.27a so that the Millington-Quirk equation for tortuosity ($Sat^{10/3}$ x $porosity^{4/3}$) can be used. Hence, the D^0 in Equation 2.27a is not the same as the D^0 in card Group 11, and differ by the tortuosity factor. Some version of the computer code needs to be checked out to verify the true required value. To complicate things, Equation 3.3 in RPP-15834 uses a D_0 notation, instead of D^0, and defines it as an effective diffusion coefficient in free water and is evidently different from the molecular diffusion coefficient in pore water, but this isn't clear in RPP-15834 as both "effective" and "molecular" are used without definitions. Equation 3.3 of RPP-15834 says that the effective diffusion coefficient, for Millington-Quirk, is $(mois)^{10/3}$/$porosity^2$ x D_0, which compares well to Equation 2.7a in VAM document if D^0 in Equation 2.7a is the same as D_0, which apparently is the diffusion coefficient of solute particles in free water. Here's the problem, the value in the input files is in effect (see 2nd paragraph above) equal to $porosity^{10/3}$ x D^*, which is the free diffusion coefficient. In spite of all of the inconsistencies in notation, I think that the VAM code is using the right values for effective diffusion, which is the same as Equation 3.3 in RPP-15834, when the index of 3.333 is chosen in Group 11 input and D^0 is set to $porosity^{4/3}$ x free diffusion coefficient in the same input line. If the linear index of 1 is chosen, instead of 3.333, then the D^0 input should be $porosity^1$ x free diffusion coefficient.

In spite of the above documentation inconsistencies, the correct molecular diffusion is probably used in the calculations for Millington-Quirk model. However, the molecular diffusion is probably a very minor phenomenon in solute transport and not as important as the diffusivity, which is defined in Equation 2.7a in VAM document. The problem

appears to be that the diffusivity is not multiplied by the moisture content in Equation 2.7a, nor in Equation 2.21, which is an error. The VAM document should show the moisture content explicitly in Equation 2.27a so that effective diffusion stays correct. In spite of the documentation error, the important aspect is the coding itself. It may be that the diffusivity (by Scheidegger) is multiplied by the moisture content in the code, which would mean the calculations are correct. If the code does not multiply the diffusivity by moisture content (or equivalently by the product of saturation and porosity) then the calculations aren't accurate, but fortunately are conservative. In other words, the calculations would show faster transport than intended, which is conservative.

I checked all of the other input values in the files and have no other issues than stated above.